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Optimal valorization of organic-biological byproducts from the primary sector with a focus on composting

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Woord vooraf

Dit is het dan, het is af, *mijn* doctoraat, maar ik zeg beter: *ons* doctoraat. Vele mensen hebben al dan niet rechtstreeks bijgedragen aan dit onderzoek, en ik wil hen dan ook van harte bedanken.

Vier jaar geleden kreeg ik een mailtje van Johan Van Waes met de vraag of ik geïnteresseerd was in een doctoraat op het ILVO over valorisatie van reststromen via compostering. Allereerst wil ik Johan en Kristiaan Van Laecke dan ook bedanken om mij het vertrouwen te geven om dit doctoraat te starten. Naast het grote maatschappelijke belang van dit thema voor een duurzame landbouw, overtuigde de samenwerking met verschillende stakeholders, om zo daadwerkelijk iets in de praktijk te implementeren, mij om dit doctoraat tot een goed einde te brengen. Hartelijk dank aan alle stakeholders die de tijd wilden maken om dit doctoraat vooruit te helpen.

Achteraf gezien was het maken van deze doctoraatsthesis eigenlijk best vergelijkbaar met het maken van compost. Om tot een kwaliteitsvol eindresultaat te komen, had ik namelijk de hulp nodig van onder meer organische stof, droge stof, oogstresten, meetapparatuur, een verreiker, emmers om sapverliezen op te vangen, micro-organismen, een compostkeerder, een zeef, een compostdoek, een ondergrond, kwaliteitscontroles en een goed werkend landbouwbedrijf.

Vooraleerst wens ik mijn promotoren te bedanken! Bert, je fungeerde als de *organische stof* tijdens mijn doctoraat, het belangrijkste element van compost. Alles start bij de organische stof en het houdt tegelijk ook alles samen. Door de vele stakeholderbezoeken waren we soms lang onderweg, maar door de gezellige babbels ging de tijd zo voorbij. Ik wil je uit de grond van mijn hart bedanken voor alle tijd die je aan mij hebt besteed, het nalezen en verbeteren van al mijn teksten, de leuke momenten met je prachtige dochters, het sneeuwballengevecht, het papier van olifantenpoep en nog zo veel meer… Tevens was je als organische stof het perfecte tegengewicht voor de vele *droge stof* (en humor) van Bart, het andere belangrijke basisingrediënt van een goede compost. Bart, wanneer ik vast zat met mijn redeneringen, kon je mij altijd helpen. Ook aan jou bedankt voor alle input, ik heb veel van je geleerd. Maar nu mijn dikke fractie recept ontwikkeld is, mag je me eindelijk eens jouw recept verklappen, want je chocomousse is de beste ☺.

Goede compost dient ook zorgvuldig gemonitord te worden om op tijd het proces bij te sturen. Tijdens mijn doctoraat nam mijn universitaire promotor Stefaan (onder meer) de rol van *meetapparatuur* op zich, waarvoor bedankt! Je kritische blik (ik was soms te 'pro' compost) en het nalezen van mijn papers kwamen dit doctoraat zeker ten goede.

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Daarnaast kon ik ook beschikken over een uitstekende *verreiker*, die alle reststromen, in de goede verhouding, netjes op een hoop legde, waardoor het proces kon starten. Koen, ik zie je als mijn compost-vader, je hebt me rondgeleid in de wondere wereld van het composteren en ik ben je heel dankbaar dat je zo veel wijsheid met me gedeeld hebt. Ondanks dat je officieel geen promotor was, heb je zeker evenveel bijgedragen, omdat je zo gepassioneerd bent door duurzame landbouw en compost. Ook bedankt om zo gek te zijn om 's morgensvroeg mee te gaan lopen in het industriële Hamburg, maar dan konden we 's avonds extra veel eten ☺.

Eten doet mij denken aan prei, sluitkool, selder en bloemkool. Laura, we zijn op dezelfde dag een doctoraat gestart. Toen wisten we nog niet dat we dezelfde promotor zouden hebben, samen incubatietesten zouden uitvoeren en twee papers zouden schrijven over *oogstresten* van groenten. Met zo een goede vriendin mogen samenwerken, zorgde voor veel hilariteit. Ik wil je bedanken voor die leuke tijden, maar ook voor zo veel meer, zoals je altijd grappige e-mails en je lekkere en verrassende eetcombinaties!

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Graag wil ik hier ook nog enkele belangrijke mensen in de bloemetjes zetten. Elien en Katrijn (alias de B.....F…...), jullie zijn al 15 jaar mijn *ondergrond* waar ik op steun, mijn 'oerewoud', waar, onder andere, ook deze compost tot stand is gekomen. Jelke, Alix, Silke, Sanne en Laurien: merci voor de etentjes, girls-dates, Sex and the City-avonden, weekendjes, vele babbels en het onbewust verzetten van mijn gedachten. Ik kan mij geen betere vriendinnen wensen die altijd voor mij klaar staan!

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En tot slot, compost is niets waard zonder *bodem* en omgekeerd. Klaas, we leerden elkaar een klein jaar geleden onverwachts 'beter' kennen. Ondertussen hebben we al veel gereisd en dubbele schelpjes gevonden, veel swingouts (zowel vanuit open als closed position) geoefend, 300 km met een tandem gefietst, heb je mij laten proeven van je lekkere kookkunsten en hebben we al veel te uitgebreid ontbeten enz... Dank je wel voor je geduld, je hulp bij mijn doctoraat, je kunst om mij te kalmeren en te motiveren, je enthousiasme en positieve ingesteldheid die mij telkens weer doen lachen, en je grappig West-Vlaams ☺. What feels like the end is often the beginning... Misschien vinden we toch nog dat tweede muntstuk?

Summary

Local on-farm composting fits within the view of a sustainable bioeconomy and agricultural systems, in which sustainable soil management and other agro-ecological practices have an important role to play, because (1) organic-biological byproducts are valorized, (2) biomass from prior biomass processing or biomass unsuitable for other processes can be valorized (i.e., it is a technique in a sequence of processing steps), (3) its production is less dependent on non-renewable resources, (4) nutrient and material cycles are closed on a local scale, and (5) compost application contributes to soil quality and fertility. However, despite these advantages, both composting and compost application are not common practices in (Flemish) agriculture. Therefore, the objective of this thesis was to **(1)** identify the challenges and hindering factors to on-farm composting and the application of compost in agriculture. Those barriers were used to further refine the aims of the thesis: **(2)** increasing insight in the processes, environmental impact and application value of on-farm composting with locally available organic residues (technical objectives) and **(3)** providing tools for farmers, policymakers and other stakeholders, ultimately stimulating feasible, efficient and sustainable on-farm composting and use of high-quality compost in Flemish agriculture (socio-economic objectives). To do so, a transdisciplinary and participatory research approach was used. We bundled expertise from various scientific disciplines and actively involved stakeholders as an essential part of the project.

(1) Challenges and hindering factors to on-farm composting and use. Twenty-eight barriers were identified and categorized as either market and financial, policy and institutional, scientific and technological, or informational and behavioral barriers. This evaluation showed that especially the shortage of available woody biomass, strict legislative preconditions, considerable financial and time investment, and lack of experience and knowledge are hindering on-farm composting. At the same time, the complex regulation, manure surplus, variable availability and composition of compost, as well as transport limitations are barriers to apply compost.

(2) To help alleviating certain barriers, we executed a set of experiments with **on-farm composting of locally available organic residues**, both from vegetable and animal origin. We hence increased insight in **the potential of these (often underutilized and/or**

fast degradable) residues and the process preconditions to avoid nutrient losses and optimize compost quality.

First, co-ensiling, co-composting and anaerobic co-digestion were compared as alternative conservation and valorization options for ***N-rich vegetable crop residues***. The three valorization options can be used to process and/or store the crop residues during winter, thus, reducing the risk for N losses linked with the fast decomposition of fresh crop residues. Furthermore, by reapplying the end products to the soil, the soil quality benefits and nutrient cycles are closed on a local scale. Compost, digestate and silage have different characteristics and, when applied to soil, lead to product-specific C and N dynamics. More specifically, soil amendment of silages, that are still highly biodegradable, resulted in highest C mineralization and microbial biomass C, and temporary N immobilization. In contrast, application of mature composts led to low C mineralization and no net N mineralization or immobilization. The effect of digestate application was comparable to compost application, but higher mineral N and N_2O emissions were found. The on-farm facilities and different soil effects of processed vegetable crop residues should be taken into account when choosing the most appropriate valorization option and application in time and space.

Secondly, field storage conditions and treatment of ***cattle farmyard manure*** were studied, again with a focus on limitation of environmental impact (N losses to the soil) and optimization of the agronomic value of this manure. The treatments in our experiments differed in terms of storage method (stockpiling, extensive composting or co-composting with bulking agents) and coverage (no cover, plastic or geotextile cover). Over the different treatments, the NH_4^+-N concentrations under the piles in the 0-90 cm soil layer amounted to a maximum of 4.2% of the initial manure N content. We were able to assess the relative importance of each of the two processes. A higher mineral N content under the piles was mainly the result of direct leaching from the cattle farmyard manure to the soil on the one hand, and of elevated soil temperatures (up to 37°C) under the piles on the other hand. NH_4^+-N was the most important component of mineral N under all heaps due to limited oxygen diffusion to the soil. N leaching and end product quality were affected by a combination of treatment option (i.e., storage and cover) and initial manure characteristics. For cattle farmyard manure with a relatively low volumetric moisture content and high C/N ratio (25 in our experiments), composting appeared to be the better technique to have the least N leaching and most stable end product. This is often the case for manure with a high straw content or with added bulking

agents. In contrast, cattle farmyard manure with a high volumetric moisture content and low C/N ratio, stockpiling and covering (plastic or geotextile) appeared to be the better technique to have the least N leaching to the soil.

A third byproduct studied was the ***solid fraction*** of dairy cattle slurry. The influence of composting or ensiling the solid fraction on the product quality in terms of fertilizer value, sanitation and stability was investigated. On-farm windrow composting of pure solid fraction was evaluated as not convenient due to oxygen shortage. Co-composting solid fraction with bulking agents led to higher pile temperatures (better sanitation), but similar compost stability as composting pure solid fraction. Despite the high temperatures, co-composting solid fraction with straw and grass did not increase spores of extreme thermophilic aerobic spore-forming bacteria, important for use as bedding material. Due to the sanitizing effect during composting and the higher N/P and C/P ratios compared to silages, composts are more favorable for export. Co-ensiling allowed for storing solid fraction without further decomposition or N losses, maintaining the fertilizer value through conservation of organic matter and nutrients. Therefore silages are relatively more suited for local applications.

When composting N-rich byproducts with a high moisture content, more porous and C-rich feedstock materials should be added. However, farmers experience a shortage in the latter type of feedstock. Therefore, we investigated the potential of three C-rich byproducts (chopped heath biomass and spent growth media of strawberry and tomato cultivation) as ***alternative bulking agents*** for wood chips during compost production. We concluded that chopped heath biomass and spent growth media can be used as bulking agent for composting green crop residues. Use of all bulking agents resulted in stable composts with an organic matter content suitable for use as soil amendment. Chopped heath biomass is a suitable alternative for wood chips and resulted in a compost with a low nutrient content and high C/P ratio (232), valuable to increase soil organic C content in P-rich soils. Spent substrates can replace wood chips, however, due to their dense structure and lower biodegradation potential, adding a coarser bulking agent is required. We suggested to use a mixture of coarse and fine bulking agents to optimize the feedstock mixture and composting process.

(3) Providing tools for farmers, policymakers and other stakeholders to stimulate on-farm composting and use.

One of the most important conclusions of part (1) of this thesis was that alternative, collaborative forms of on-farm composting might function as a potential lever to overcome some of the most stringent legislative, market and financial barriers. To gain more insights in the feasibility of some of these alternative production forms, three action research case studies were performed on different locations in Flanders, in collaboration with stakeholders and relevant policy advisors and implementers. Compost quality, production costs and applicable regulation per case were monitored. We examined two types of collaboration. In the first type, the on-farm production process was outsourced to a service provider (contractor), who was responsible for the set-up and monitoring of the on-farm composting process. In the second type, on-farm composting was performed by the collaboration between different farmers and nature conservationists, exchanging biomass and compost. The cases studies indicated that cooperation between different partners for the production of on-farm compost resulted in a better and economically more feasible composting process. Further, five general factors determining the choice of compost production set-up and costs were illustrated: (1) available biomass, (2) local opportunities, (3) local compost requirements, (4) agreements made between partners and (5) facilities and capabilities of the partners. In conclusion, we formulated a set of suggestions for adjustments in legislation and policy, which could enhance chances for on-farm composting and farm compost application.

In conclusion, this thesis clearly showed opportunities for on-farm composting and compost application in Flemish agriculture. Future research should focus on the evaluation of nutrient losses (to air and water), costs and environmental benefits for different management practices throughout the entire management chain: from stable or field, over storage, spreading on the field and further decomposition in the soil.

Samenvatting

Boerderijcomposteren past in de visie van een duurzame bio-economie en landbouwsystemen waar duurzaam bodembeheer en andere agro-ecologische praktijken een belangrijke rol spelen, omdat (1) organisch-biologische restproducten gevaloriseerd worden, (2) resterende biomassa uit andere valorisatieprocessen of biomassa dat niet geschikt is voor hoogwaardige valorisatie, kan gevaloriseerd worden, (3) de productie minder afhankelijk is van niet-hernieuwbare hulpbronnen, (4) nutriënten- en materiaalkringlopen gesloten worden op lokaal niveau, en (5) composttoepassing bijdraagt tot bodemkwaliteit en -vruchtbaarheid. Ondanks deze voordelen, zijn composteren en toepassing van compost geen algemeen voorkomende prakijken in de (Vlaamse) landbouw. Vandaar is het doel van dit PhD onderzoek **(1)** het identificeren van uitdagingen en hinderende factoren van boerderijcomposteren en composttoepassing in de landbouw. Deze knelpunten werden gebruikt om de onderzoeksdoelstellingen verder te verfijnen: **(2)** meer inzicht verwerven in de processen, milieu-impact en landbouwkundige waarde van boerderijcomposteren met lokaal beschikbare organische reststromen (technische doelstellingen) en **(3)** aanreiken van hulpmiddelen voor landbouwers, beleidsmakers en andere stakeholders, om uiteindelijk haalbare, efficiënte en duurzame boerderijcompostering en toepassing van kwalitatieve compost in de Vlaamse landbouw te stimuleren (socio-economische doelstellingen). Een transdisciplinaire en participatorische onderzoeksaanpak werd gebruikt, expertise van verschillende wetenschappelijke disciplines werd gebundeld en stakeholders werden actief betrokken in het project.

(1) Uitdagingen en hinderende factoren voor boerderijcomposteren en compostgebruik. Achtentwintig knelpunten werden geïdentificeerd en gecategoriseerd als ofwel markt- en financieel gerelateerde knelpunten, barrières in het beleid en de wetgeving, wetenschappelijke en technologische knelpunten of informationele en gedragsgerelateerde knelpunten. Uit deze evaluatie bleek dat vooral het tekort aan beschikbare houtige reststromen, de strikte regelgevende vereisten, aanzienlijke financiële en tijdsinvestering en het gebrek aan ervaring en kennis boerderijcomposteren moeilijk maken. Tegelijkertijd wordt de toepassing van compost in de landbouw vooral tegengehouden door de complexe wetgeving, het mestoverschot, de variabele beschikbaarheid en samenstelling van compost en het moeilijke transport.

(2) Om bepaalde knelpunten te helpen opheffen werden verschillende experimenten uitgevoerd waarbij **lokaal beschikbare organische reststromen, zowel plantaardig als dierlijk van oorsprong, werden gecomposteerd**. Daardoor werd inzicht verworven in het **potentieel van deze (vaak onderbenutte en/of snel degradeerbare) reststromen en de randvoorwaarden van de processen om nutriëntenverliezen te vermijden en compostkwaliteit te optimaliseren**.

Eerst werden co-inkuilen, co-composteren en anaerobe co-vergisting vergeleken als alternatieve bewaar- en valorisatieopties voor ***N-rijke oogstresten van groenten***. De drie valorisatieopties kunnen gebruikt worden om oogstresten te verwerken en/of op te slaan tijdens de winter, waardoor het risico op N verliezen, gelinkt aan de snelle afbraak van verse gewasresten, gereduceerd wordt. Het terug toepassen van de eindproducten op de bodem heeft voordelen voor de bodemkwaliteit en sluit nutriëntenkringlopen op lokaal niveau. Compost, digestaat en kuil hebben verschillende eigenschappen en leiden tot productspecifieke C en N dynamieken wanneer ze worden toegepast. Meer specifiek, bodemtoepassing van kuilen, die nog zeer biodegradeerbaar zijn, leidde tot de hoogste C mineralisatie en microbiële biomassa, en tijdelijke N immobilisatie. Toepassing van stabiele composten resulteerde daarentegen in lage C mineralisatie en geen netto N mineralisatie of immobilisatie. Het effect van digestaattoepassing was vergelijkbaar met composttoepassing, maar hogere minerale N en N_2O emissies werden gemeten. De faciliteiten op het bedrijf en de verschillende effecten van het toepassen van de verwerkte gewasresten op de bodem moeten in rekening worden gebracht om de meest aangewezen valorisatieoptie en toepassing in tijd en ruimte te kiezen.

Ten tweede werd de opslag en behandeling van ***runderstalmest*** op het veld bestudeerd, opnieuw met de focus op het reduceren van de milieu-impact (N verliezen naar de bodem) en de optimalisatie van de agronomische waarde van de mest. De behandelingen in onze proeven varieerden in opslagmethode (los storten, extensief composteren of co-composteren met bulkmaterialen) en afdekking (geen afdekking, plastic doek of geotextiel). Over de verschillende behandelingen heen bedroeg de NH_4^+-N concentratie onder de hopen in de 0-90 cm bodemlaag maximum 4.2% van de initiële N inhoud van de mest. Deze verhoogde N concentratie was vooral te wijten aan directe uitspoeling van N uit de mesthopen naar de bodem, maar er was ook een kleiner effect van verhoogde bodemtemperaturen (tot 37°C) onder de hopen die leidden tot verhoogde N mineralisatie in de toplaag. NH_4^+-N was de belangrijkste component van minerale N onder alle hopen, te verklaren door de beperkte diffusie van zuurstof in de bodem. N

uitspoeling en productkwaliteit werden beïnvloed door een combinatie van behandeling (opslag en afdekking) en initiële mestkarakteristieken. Voor runderstalmest met een relatief laag volumetrisch vochtgehalte en hoge C/N verhouding (25 in onze experimenten) leidde composteren tot de minste N uitspoeling en het meest stabiele eindproduct. Dit is vaak het geval voor mest met een hoog strogehalte of wanneer bulkmaterialen worden bijgemengd. Runderstalmest met een hoog volumetrisch vochtgehalte en lage C/N verhouding wordt het best los gestort en afgedekt (met een plastic of geotextiel) om N uitspoeling naar de bodem te minimaliseren.

Een derde reststroom die onderzocht werd is de ***dikke fractie*** na scheiden van runderdrijfmest. De invloed van composteren of inkuilen op de productkwaliteit, meer bepaald de bemestende waarde, hygiënisatie en stabiliteit werd bestudeerd. Boerderijcomposteren van pure dikke fractie was niet gunstig door zuurstofgebrek. Co-composteren met bulkmaterialen leidde tot hogere temperaturen in de hopen (betere hygiënisatie), maar resulteerde in eenzelfde productstabiliteit dan na composteren van pure dikke fractie. Ondanks de hoge temperaturen leidde co-composteren van dikke fractie met een mengsel van stro en gras niet tot een verhoogd risico op sporen van extreem thermofiele aerobe sporenvormende bacteriën, een belangrijk criterium voor toepassing van dikke fractie als beddingmateriaal. Het hygiëniserende effect van composteren en de hogere N/P en C/P verhoudingen van de composten in vergelijking met de kuilen, zorgen er voor dat compost meer aangewezen is voor export. Co-inkuilen laat opslag van dikke fractie toe zonder verdere afbraak of N verliezen en met behoud van de bemestende waarde (behoud van organische stof en nutriënten). Vandaar zijn kuilen relatief meer gepast voor lokale toepassingen.

Bij het composteren van natte, N-rijke reststromen is het noodzakelijk om meer poreuze en C-rijke materialen toe te voegen. Desondanks ervaren veel landbouwers een tekort in deze C-rijke stromen. Daarom onderzochten we het potentieel van drie C-rijke reststromen (heidechopper en gebruikte groeimedia van aardbei- en tomatenteelt) als ***alternatieve bulkmaterialen*** voor houtsnippers in boerderijcomposteren. We besloten dat heidechopper en groeimedia gebruikt kunnen worden als bulkmateriaal voor het composteren van N-rijke oogstresten. Co-composteren met deze bulkmaterialen resulteerde in stabiele composten met een organische stofgehalte geschikt voor bodemverbetering. Het gebruik van heidechopper zorgde voor een compost met een lage nutriënteninhoud en een hoge C/P verhouding (232), zeer waardevol om het bodem

organische stofgehalte in P-rijke bodems te verhogen. Gebruikte groeimedia kunnen evenzeer houtsnippers vervangen, maar door hun fijne structuur en lage biodegradeerbaarheid is het nodig om een meer poreus bulkmateriaal toe te voegen. Een mengsel van grove en fijne bulkmaterialen lijkt het meest aangewezen om het uitgangsmengsel en het composteerproces te optimaliseren.

(3) Aanreiken van hulpmiddelen voor landbouwers, beleidsmakers en andere stakeholders om boerderijcomposteren en compostgebruik te stimuleren.

Een van de belangrijkste conclusies van deel (1) van dit onderzoek was dat alternatieve, collaboratieve vormen van boerderijcomposteren kunnen dienen als hefboom om sommige wetgevende, marktgerelateerde en financiële knelpunten te overwinnen. Daarom werd de haalbaarheid van sommige van deze alternatieve productievormen getest via het uitvoeren van drie case studies op verschillende locaties in Vlaanderen, in samenwerking met stakeholders en relevante beleidsadviseurs en -uitvoerders. Compostkwaliteit, productiekosten en regelgeving per case werden in kaart gebracht tijdens dit actieonderzoek. In het eerste type samenwerkingsvorm werd het productieproces op het landbouwbedrijf uitbesteed aan een dienstverlener (loonwerker), verantwoordelijk voor de opzet en opvolging van het composteerproces. In het tweede type werd het boerderijcomposteren uitgevoerd via een samenwerking tussen landbouwers en natuurbeheerders die biomassa en compost uitwisselden. We besloten dat samenwerking tussen verschillende partners voor het produceren van boerderijcompost resulteerde in een beter en economisch meer haalbaar composteerproces. De drie case studies illustreerden vijf algemene factoren die de keuze van de opzet van de compostproductie en de kosten beïnvloedden: (1) beschikbare biomassa, (2) lokale opportuniteiten, (3) lokale vereisten van de compostsamenstelling, (4) gemaakte afspraken tussen de partners en (5) faciliteiten en mogelijkheden van de partners. Tot slot formuleerden we een aantal suggesties tot aanpassingen in de regelgeving en het beleid die de kansen voor boerderijcomposteren en composttoepassing kunnen verhogen.

In het algemeen toonde dit PhD onderzoek verschillende opportuniteiten om boerderijcomposteren en composttoepassing in de Vlaamse landbouw te stimuleren. Het is echter noodzakelijk om nog onderzoek uit te voeren dat focust op de evaluatie van nutriëntenverliezen, kosten en milieuvoordelen tijdens verschillende valorisatieopties, rekening houdende met de hele keten: van stal of veld over opslag, uitspreiden op het veld en verdere afbraak in de bodem.

List of abbreviations

AD	anaerobic digestion
ADF	acid detergent fiber
ADL	acid detergent lignin
ANOVA	analysis of variance
BA	bulking agent
BAU	business as usual
C	carbon
CFM	cattle farmyard manure
$C_{min, net}$	net C mineralization
DM	dry matter
EC	electrical conductivity
FW	fresh weight
FYM	farmyard manure
MAP	manure action plan
MBC	microbial biomass C
n	number of replicates
N	nitrogen
nda	no data available
NDF	neutral detergent fiber
NFRV	nitrogen fertilizer replacement value
NH_4^+-N	ammonium-N
$N_{min, net}$	net N mineralization
NO_3^--N	nitrate-N
OM	organic matter
OUR	oxygen uptake rate
SF	solid fraction
SOC	soil organic carbon
TOC	total organic carbon

Overview of the treatments

Chapter 3

Trial 1

CON1 and $CON1_{60N}$	Control soil (0N) and control soil + 60N
CC1	CON + composted cabbage
LC1	CON + composted leek
CS1 and $CS1_{60N}$	CON + ensilaged cabbage and CON + ensilaged cabbage + 60N
LS1 and $LS1_{60N}$	CON + ensilaged leek and CON + ensilaged leek + 60N
CD1	CON + digested cabbage
LD1	CON + digested leek

Trial 2

CON2 and $CON2_{60N}$	Control soil (0N) and control soil + 60N
FL	CON + fresh leek
$LC2_{heath}$	CON + composted leek/heath chopper
$LC2_{strawberry}$	CON + composted leek/strawberry substrate
LS2 and $LS2_{60N}$	CON + ensilaged leek and CON + ensilaged leek + 60N
LD2	CON + digested leek

Chapter 4

Cw	Co-composting leek residues with wood chips
Ch	Co-composting leek residues with chopped heath biomass
Cs	Co-composting leek residues with spent strawberry substrate
Ct	Co-composting leek residues with spent tomato substrate

Chapter 5

Experiment 1

S1	Uncovered stockpiling of CFM
C1	Composting of CFM

Experiment 2

S2	Uncovered stockpiling of CFM
ST2	Covered storage with TopTex of CFM
SP2	Covered storage with plastic of CFM
C2	Composting of CFM

Experiment 3

C3	Composting of CFM
CG3	Co-composting of CFM with old grass clippings
CC3	Co-composting of CFM with pre-composted old grass clippings

Chapter 6

Field study

SF_C	Composting SF
SF+Clin_C	Co-composting SF with clinoptilolite
SF+CFM_C	Co-composting SF with CFM
SF+S+G_C	Co-composting SF with straw and grass
SF+S+G_E	Co-ensiling SF with straw and grass

Lab study

SF_E_{lab}	Ensiling SF
$SF+Clin\_E_{lab}$	Co-ensiling SF with clinoptilolite
$SF+CFM\_E_{lab}$	Co-ensiling SF with CFM
$SF+S+G\_E_{lab}$	Co-ensiling SF with straw and grass

List of tables

List of figures

Table of contents

CHAPTER 1

Introduction

Increasing population, scarcity of resources and materials, environmental pressures (e.g. pollution), and climate change are issues that challenge our current society and our agriculture and agro-food system. At the same time, one-third of the global food produced for human consumption is lost (about 1.3 billion tonnes per year according to Gustavsson et al. (2011)), while on European level around 88 million tonnes of food is wasted annually (Stenmarck et al., 2016). Moreover, current food and feed production generally results in the valorization of only a part of the total biomass available, while the remaining flow of byproducts is mainly used for low-value applications, incinerated or dumped. To help address these issues, policy makers are taking initiatives to reduce food losses (e.g. establishment of the *EU Platform on Food Losses and Food waste*, with a target to halve food waste at the retail and consumer level by 2030 and *Ketenroadmap Voedselverlies 2020*, a Flemish public-private agreement to decrease food losses by 15% by 2020).

Besides and strongly related to this issue, policy makers (e.g. European Commission, 2012; The White House, 2012) are also encouraging rapid development of the **bioeconomy**, which relies on renewable biomass instead of finite fossil inputs for the production of value-added products such as food, feed, biobased products and bioenergy (OECD, 2013). Furthermore, to maximally valorize the available biomass, these products should be produced according to a cascade principle, meaning that biomass is first processed into high-value products (e.g. food, feed, chemicals), after which the residues may be used for lower value applications (e.g. energy) (OECD, 2013). Hence, the bioeconomy encompasses both the activities related to the production of biomass and the different ways by which this biomass and its byproducts are used. The main products on which the bioeconomy is based, are derived from agricultural activities. Consequently, also within the agricultural and agro-food industries more and more attention is paid to efficient use of basic resources, including (food) waste reduction and better closure of the cycles of resource use. To move from our current fossil-based economy to a sustainable and competitive bioeconomy, a transition is necessary.

As such, also in Flanders, the Flemish government has developed a strategic vision and framework[1], as a result of a joined effort of the interministerial working group in which entities of different policy domains are actively involved. According to this strategy, a bioeconomy will offer huge opportunities for, among other things, the further development of a circular economy, the optimal use of raw materials and economic growth. Recently, several research and action programs related to this hot topic were initiated in Flanders. Some examples are the '*VISIONS project*' aiming to identify the main organic waste streams and byproducts in Flanders with the ambition to use these products in new value chains; '*FISCH*', Flanders Innovation Hub for Sustainable Chemistry striving for the realization of new value chains based on the application of sustainable chemistry; and the *GENESYS* project, aiming at system innovation for the valorization of agro-food and fisheries byproducts. This thesis is embedded in the latter project, the **GENESYS project** funded by the Institute for Agricultural and Fisheries Research of Flanders, Belgium (ILVO).

The GENESYS project and its approach, an overview of the research objectives and an outline of the thesis, followed by a brief description of the basic concepts of the composting process are presented in this first introductory chapter.

1.1. The GENESYS project - Innovative pathways for the valorization of agro-food and fisheries byproducts through a 'different', systemic approach

The GENESYS project[2] combines a thematic and a methodological objective, realized through four parallel PhD research trajectories. The thematic objective is to **optimize the valorization of animal and plant-based byproducts through innovation**, and is realized through three specific case studies. In a first case, focus is on the valorization of vegetable byproducts, more specifically tomato and chicory as model crops, into high-value applications for food, feed and chemistry. In a second case, byproducts comprising of low-value bycatches and discards from the fisheries industry are valorized through ensiling, after which the end products can be used in the aquafeed industry. In a third thematic case, which is the topic of this thesis, organic byproducts (both from vegetable and animal sources) are

[1] https://www.vlaanderen.be/nl/publicaties/detail/bioeconomy-in-flanders

[2] http://www.ilvogenesys.be/

valorized through composting (and alternatively ensiling), in order to produce soil improvers and fertilizers, thereby closing nutrient and material cycles and investing in agricultural soil quality.

However, since classic science-driven innovation research often hinders the end-users from adopting the innovation in practice, the methodological objective is to devise instruments that support the development of **successful system innovations**. Hence, in the thematic cases a methodological innovative research approach (Van Lancker et al., 2016) was used, that targets the implementation of the innovation by the end users. While classic science-driven innovation research is often restricted to a single discipline and follows a rather rigid linear approach, we aimed at performing **transdisciplinary research in an innovative way**. We bundled expertise present in various scientific disciplines and actively involved stakeholders (e.g. farmers, supply chain members, technologists, scientists and policy makers) as an essential part of the project. The initially gathered information and challenges (see Chapter 2) were used to further refine the research questions and approach of the thesis. The stakeholders remained involved throughout the entire project to expose problems, challenges and possible solutions from a scientific and practical viewpoint. This stakeholder participation enabled to find scientifically supported and socially acceptable solutions for valorizing agro-food byproducts.

1.2. What is compost and how does it fit in this story?

The increasing importance of the bioeconomy requires a sufficiently high and continuous availability of biomass as feedstock for biobased product development, and thus also a reliably high level of biomass production resources, also called natural capital, including soil, water and biodiversity. One of the most important resources required to grow biomass, is a fertile and healthy soil (Meyer-Kohlstock et al., 2013). Improving and maintaining soil quality and fertility in a sustainable way is an important challenge for modern agriculture, since input-intensive agricultural practices such as the use of mineral fertilizers, frequent and intensive soil tillage, narrow crop rotations, and past shifts in land use (Sleutel et al., 2003; Sleutel et al., 2006) have led to decreased soil organic carbon (SOC) stocks, biodiversity loss, soil erosion, and pollution of groundwater and air (Kirschenmann, 2010). Furthermore, those unsustainable agricultural practices do not fit within the concept of the bioeconomy, in which resources are managed and used in a sustainable way, guaranteeing their use for the long term. This management also includes a more efficient use of nutrients such as N, P,

K and trace elements, by closing nutrient cycles on different scale levels, preventing exhaustion of those nutrients and water pollution by excess use of them.

Soil quality and fertility entails a sufficiently high level of SOC, since this is one of the most important indicators of soil fertility (Aggelides and Londra, 2000; Stamatiadis et al., 1999; Turner et al., 1994; Zebarth et al., 1999; D'Hose et al., 2014). To sustainably increase SOC following the bioeconomy vision, alternative management strategies are needed, and can include alterations in crop rotation, rotation with temporary grassland, reduced soil tillage, use of (leguminous) green manure crops (cover crops) and the use of organic fertilizers. Examples of such organic fertilizers are farmyard manure, slurry, cut-and-carry fertilizers, digestate and compost. Composting and the usage of compost have several specific characteristics that make it fit particularly well within the logics of the bioeconomy and circular economy.

Composting is a biological process during which microorganisms convert fresh organic material into a stable and humus-rich product under controlled conditions, i.e., optimal conditions of moisture and oxygen to facilitate the decomposition process (Bokhorst and ter Berg, 2001; Willekens and Cloet, 2003). Besides the advantages of compost application on soil quality, compost can play an important role in the bioeconomy because its production is less dependent on non-renewable resources. Compost can be produced locally on farm (further referred to as on-farm composting), and can use biomass that is not used further in other biomass cascade applications (e.g. materials not suitable for digestion or combustion) as well as byproducts from other applications (e.g. digestate remaining after bioenergy production) as feedstock. This limits the competition for biomass, as composting is a process that can be included (typically at the end) in various biomass cascades to reduce the volume and moisture content as well as stabilize and sanitize the biomass. Furthermore, compost application on agricultural soils closes organic carbon and nutrient cycles, thus contributing to sustainable soil fertility and fertilization.

About 15 years ago, some individual Flemish farmers started with on-farm composting (windrow composting system). From 2003 till 2007, ILVO managed a consultancy service FarmCOMPOST, financed by IWT, a Flemish government agency for Innovation by Science and Technology, and SYMBIOS, a farmer organization. FarmCOMPOST advisors assisted on-farm composting and were advising about sustainable soil management. More recently,

Bioforum, the Flemish umbrella organization of the organic sector, promoted on-farm composting by publishing a brochure about composting techniques and related legislation. Despite those efforts for knowledge transfer and the global awareness of the benefits of compost amendment, compost production on farm and compost application is not a common practice in Flanders.

1.3. Objectives and outline of the thesis

The overall aim of this thesis is stimulating feasible, efficient and sustainable on-farm composting and use of high-quality compost in Flemish agriculture. This is achieved by three objectives: **(1)** identification of the challenges and hindering factors to on-farm composting and the application of compost in agriculture. Following the GENESYS approach, those challenges and hindering factors were used to further refine the objectives of the thesis: **(2)** increasing insight in the processes, environmental impact and application value of on-farm composting with locally available organic residues (technical objectives) and **(3)** providing tools for farmers, policymakers and other stakeholders (socio-economic objectives) to stimulate on-farm composting in practice.

The technical and socio-economic challenges and hindering factors are shown in Figure 1.1. A first important challenge is **the optimization of composting techniques for 'difficult', underutilized and/or fast degradable farm residues.** This issue is studied in Chapters 3 (vegetable crop residues), 4 (leek residues), 5 (cattle farmyard manure) and 6 (solid fraction of cattle slurry), and the composting techniques are compared with other valorization options, i.e. ensiling and anaerobic digestion. Chapter 4 furthermore also deals with a second challenge, i.e. the frequently experienced **shortage of C-rich feedstock materials** (such as wood chips, bark or straw) in composting. Thereto, we assessed the potential use of byproducts from horticulture and nature management as alternative C-rich bulking agents. Environmental issues, and more particularly **the risk for N losses** to the soil, are studied in Chapter 3 (by evaluating alternative processing techniques instead of disposal of/leaving N-rich crop residues on the field) and Chapter 5 (by screening different storage treatments and conditions for cattle farmyard manure on the field). **Quality assessment of composts**, i.e. the characterization of their physico-chemical composition and their effect on N and C dynamics after soil application, are mainly discussed in Chapters 3 (for composted, ensilaged

and digested crop residues) and 6 (for composted and co-composted solid fraction of cattle slurry). Stability indicators for composts and silages are evaluated in Chapters 3, 4, 5 and 6.

Besides the abovementioned rather technical issues, we also focused on other challenges related to the feasibility of on-farm composting, i.e. dealing with legislative aspects, logistics, time and financial investments. Thereto, in Chapter 7, we discuss the results of a set of case studies in which **alternative and collaborative organization forms of composting** have been tested on farm.

Finally, in Chapter 8, the global results and general conclusions are presented, and suggestions for future research are outlined. A schematic overview of the thesis is illustrated in Figure 1.2.

Figure 1.1 Challenges and hindering factors for on-farm composting (green boxes).

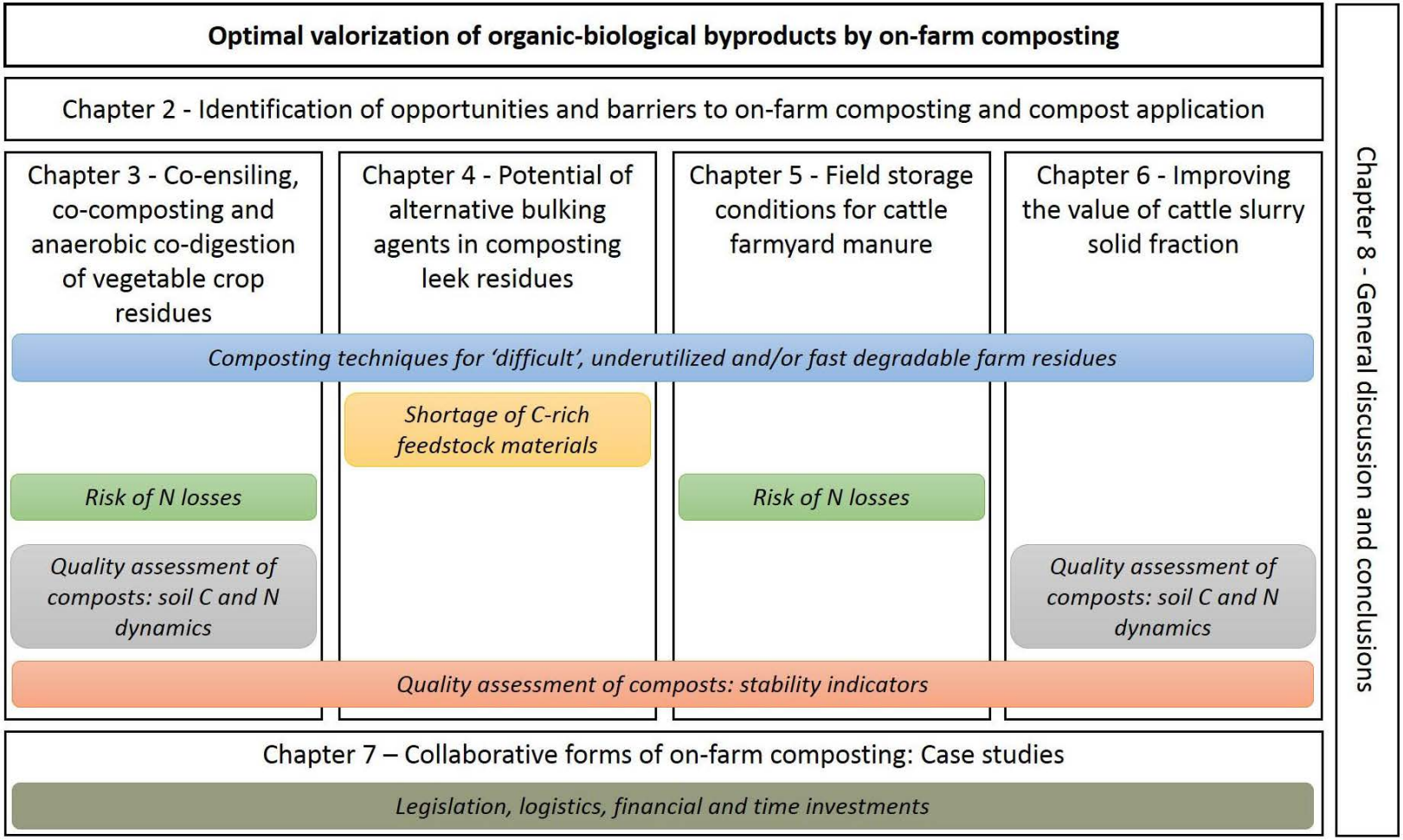

Figure 1.2 Thesis outline. Challenges and hindering factors of on-farm composting dealt with in the different chapters are marked italic in colored bars.

1.4. The basic concepts of the composting process

To obtain a high quality compost, the composting conditions should be optimal to stimulate the aerobic microorganisms to decompose the organic matter and turn it into a humus-rich product. The composting process is affected by the feedstock materials and composition and by the composting circumstances. By monitoring the process, it can be adjusted if necessary. The basic characteristics of an on-farm windrow composting system, as performed during the experiments in this study, are summarized in this section.

1.4.1. Setting-up the composting pile

Basically, the aerobic microorganisms need an energy source (C-rich organic material), nutrients (e.g., N and P), water and oxygen. The feedstock mixture at the start should have a **C/N ratio** between 25 and 35, because it is considered that the microorganisms require 30 units of C per unit of N (Tuomela et al., 2000). Higher C/N ratios slow down the decomposition as there is a shortage of N, whereas an excess of N (lower C/N ratios) can result in N losses (Bernal et al., 2009). Further, also the nature and accessibility of the C is important, because the C should be available for the microorganisms (e.g., C in lignin-rich materials is less degradable) (Van der Wurff et al., 2016). To increase the C/N ratio of a mixture of N-rich residues, bulking agents rich in C can be added. We use the term *co-composting* when referring to a composting process where additional feedstock materials have been added to an original feedstock source. Bulking agents are also important to increase the porosity of the pile, increasing gas exchange and so enhancing oxygen availability (Barrington et al., 2002). As a rule-of-thumb, it is suggested to mix 40% green (more N-rich) residues with 60% brown (more C-rich residues) on a volume basis (Bokhorst and ter Berg, 2001). The separate feedstock materials are put on a windrow (in layers) of approximately 1.5-2 m high and 3-4 m wide (Van der Wurff et al., 2016).

1.4.2. Managing the process

The piles are turned to mix the materials and to maintain aerobic conditions. This is usually and ideally done using a compost turner, but in case such turner is not available, a front loader and manure spreader can be used. Optimum **moisture content** levels at the start (50-60%) and during composting are reached by watering while turning the piles (Bernal et al., 2009). After mixing, pile temperatures will increase quickly, which is indicative for microbial activity and is in favor of the destruction of weeds and pathogens (Bokhorst and ter Berg, 2001). The range of 52-60°C is the most favorable for decomposition. At too high

temperatures, microbial activity declines rapidly (Miller, 1992). However, due to this intense microbial activity or in case of a low porosity of the mixture, **oxygen concentrations** can decrease. Similarly, due to elevated temperatures in the pile, moisture losses by evaporation will occur. Therefore, monitoring temperature, oxygen concentrations (or CO_2 concentrations as an indicator for oxygen availability) and moisture content is necessary. When the critical temperature and/or CO_2 concentrations (65°C and/or 16% CO_2) are reached, it is necessary to aerate the piles by turning. This also enables controlling the temperature, removing excess CO_2 and/or bringing in new oxygen in order to ensure optimal microbial activity. To prevent the infiltration of rainwater and nutrient losses by percolation, it is suggested to cover the pile with a geotextile cover, that allows gas exchange.

1.4.3. End of process and characterization of the final product

Due to CO_2 and water losses, the organic and dry matter content and C/N ratio gradually decrease during composting. The remaining C is in a more stabilized form. With an intensive on-farm composting process, a stabilized end product can be obtained after 8-12 weeks, as indicated by e.g. a pile temperature around the ambient air temperature, a NO_3^--N/NH_4^+-N ratio > 1 and an oxygen uptake rate < 15 mmol kg^{-1} OM h^{-1}. To have an indication about the compost quality, samples are taken and analyzed (Table 1.1), and compared with the composition of Federal and *Vlaco* (the Flemish compost organization) standards for compost, and average composition of green waste compost (GWC) and vegetable, fruit and garden waste (VFG) compost of *Vlaco* (Table 1.2). For the majority of the parameters, the analyses are performed on the fraction < 10 mm, because in industrial compost plants, the compost is sieved and only the fine fraction is sold. The sieve overflow is then reused as bulking agent in a next composting batch. To have an idea about the effect of compost amendment on soil C and N dynamics and crop growth, incubation and pot experiments can be performed.

Table 1.1 Commonly analyzed compost parameters and the corresponding method (European standard, EN).

Parameter	Method
Fresh bulk density (kg m^{-3})	EN 13040
Organic matter content (% of DM)	EN 13039
Dry matter content (% of fresh weight)	EN 13040
Total N (g kg^{-1} DM)	Dumas EN 13654-2
Total P (g kg^{-1} DM)	Ashing and digestion with 7N HNO3 and measurement with Varian CARY 50 Spectrophotometer
pH-H_2O (-)	EN 13037
Electrical conductivity (µS cm^{-1})	EN 13038
NO_3^--N (mg L^{-1})	EN 136552, measurement with DX-600 IC ion chromatograph
NH_4^+-N (mg L^{-1})	Measurement with Skalar SAN++ flow analyzer
SO_4^{2-} (mg L^{-1})	EN 136552, measurement with DX-600 IC ion chromatograph
Cl^- (mg L^{-1})	EN 136552, measurement with DX-600 IC ion chromatograph
Oxygen uptake rate (mmol kg^{-1} OM h^{-1})	Based on Grigatti et al. (2011)
Hemicellulose (% of OM)	Based on Van Soest et al. (1991)
Cellulose (% of OM)	Based on Van Soest et al. (1991)
Lignin (% of OM)	Based on Van Soest et al. (1991)
Plant-available nutrients (mg L^{-1} substrate)	Extraction in ammonium acetate (1:5 v/v) and measurement with ICP-OES
Total nutrients (mg kg^{-1} DM)	Ashing and digestion with 7N HNO_3 and measurement with ICP-OES

Table 1.2 Federal and *Vlaco* standards for compost, and average composition of green waste compost (GWC) and vegetable, fruit and garden waste (VFG) compost of *Vlaco*. OM = organic matter, DM = dry matter, FM = fresh matter.

	Federal standards	Vlaco standards	GWC Vlaco	VFG compost Vlaco
OM content (% of DM)			33	36
OM content (% of FM)	*> 16*	*> 18*	20	25
DM content (% of FM)	*> 50*	*> 55*	60	70
Total N (g kg^{-1} DM)			11.7	17.1
Total P (g kg^{-1} DM)			2.0	5.1
C/N (-)			17	12
C/P (-)			90	36
N/P (-)			5.9	3.4
NO_3^--N (mg kg^{-1} DM)				
NH_4^+-N (mg kg^{-1} DM)				
NO_3^--N / NH_4^+-N (-)		*< 10*		
Fresh bulk density (kg m^{-3})				
Germinated weeds (number L^{-1})	*max. 1*	*< 1*	0	0
pH-H_2O (-)	*6.5 - 9.5*		8	8
Electrical conductivity (µS cm^{-1})			1000	2500
Oxygen uptake rate (mmol kg^{-1} OM h^{-1})	*< 15*	*< 10*		
K (g kg^{-1} DM)			8.3	13.8

CHAPTER 2

Opportunities and barriers

After: Viaene, J., Van Lancker, J., Vandecasteele, B., Willekens, K., Bijttebier, J., Ruysschaert, G., De Neve, S., Reubens, B., 2016. Opportunities and barriers to on-farm composting and compost application: A case study from northwestern Europe. Waste Management 48, 181-192.

Abstract

Maintaining and increasing soil quality and fertility in a sustainable way is an important challenge for modern agriculture. The burgeoning bioeconomy is likely to put further pressure on soil resources unless they are managed carefully. Compost has the potential to be an effective soil improver because of its multiple beneficial effects on soil quality. Additionally, it fits within the bioeconomy vision because it can valorize biomass from prior biomass processing or valorize biomass unsuitable for other processes. However, compost is rarely used in intensive agriculture, especially in regions with high manure surpluses. The aim of this research is to identify the barriers to on-farm composting and the application of compost in agriculture, using a mixed method approach for the case of Flanders. The significance of the 28 identified barriers is analyzed and they are categorized as market and financial, policy and institutional, scientific and technological and informational and behavioral barriers. More specifically, the shortage of woody biomass, strict regulation, considerable financial and time investment, and lack of experience and knowledge are hindering on-farm composting. The complex regulation, manure surplus, variable availability and transport of compost and variable compost quality and composition are barriers to apply compost. In conclusion, five recommendations are suggested that could alleviate certain hindering factors and thus increase attractiveness of compost use in agriculture.

2.1. Introduction

Input-intensive agricultural practices such as the use of mineral fertilizers, frequent soil tillage, narrow crop rotations, and past shifts in land use (Sleutel et al., 2003; Sleutel et al., 2006) have led to decreased soil organic carbon (SOC) stocks, biodiversity loss, soil erosion, and pollution of groundwater and air (Kirschenmann, 2010). One of the most important characteristics of soil fertility is sufficient SOC (Aggelides and Londra, 2000; Stamatiadis et al., 1999; Turner et al., 1994; Zebarth et al., 1999), while the SOC content of many croplands in temperate regions is declining (European Commission, 2006; Maes et al., 2012; Sleutel et al., 2003; Van-Camp et al., 2004). Improving and maintaining soil quality and fertility in a sustainable way is thus an important challenge for modern agriculture. Moreover, policymakers (e.g. European Commission, 2012; The White House, 2012) are encouraging rapid development of the bioeconomy, which relies on renewable biomass instead of finite fossil inputs for the production of value-added products such as food, feed, biobased products and bioenergy (OECD, 2013). As a consequence, this development will require a high soil fertility and increases the need for sustainable soil improvers, since fertile soils are the prerequisite to reliably produce the necessary biomass as feedstock for food and biobased products (Meyer-Kohlstock et al., 2013).

Compost application has well-established beneficial impacts on soil quality, soil fertility and the environment. Despite knowledge of these benefits, compost application and compost production on the farm (referred to below as *on-farm composting*) is not a common practice in Flanders (the northern region of Belgium), a region characterized by large manure surpluses. We have evaluated the current challenges regarding on-farm composting and compost application in Flemish agriculture in the context of sustainable soil management. In this paper we (1) critically review the potential strengths of compost application, and (2) describe the current compost production and use in Flanders. Next, (3) we analyze the barriers to on-farm composting and compost application, and based on this analysis (4) we formulate a number of preliminary recommendations to alleviate certain barriers. A mixed method approach was used to analyze the case of Flanders, which can be exemplary to other northwestern European regions, in particular Denmark, The Netherlands, North-West Germany, the North-West of France, the Po-valley in Italy and parts of England, which have a similar climate and intensity of agriculture (Sleutel et al., 2007), and the same problems with water quality (Velthof et al., 2014).

2.2. Review of the characteristics and potential strengths of compost application

To sustainably increase SOC, farmers should change their soil management practices that often rely heavily on the application of mineral fertilizers and intensive soil tillage. Alternative management strategies for increasing or maintaining SOC can include alterations in crop rotation, rotation with temporary grassland, reduced soil tillage, use of (leguminous) green manure crops (cover crops) or the use of organic fertilizers. Examples of such organic fertilizers are farmyard manure, slurry, cut-and-carry fertilizers, digestate and compost. This study analyses on-farm compost use and production because compost has a number of extra benefits compared to other fertilizers and soil improving agents. However, compost application can also be associated with a number of drawbacks such as the risk for greenhouse gas emissions during production (Hao et al., 2001). Furthermore, the benefits and drawbacks might be influenced by climate, soil type, crop succession, feedstock mixture, compost dose, etc. Moreover, the benefits and drawbacks are depending on whether compost replaces other fertilizers or compost is used in addition to fresh manure, organic or mineral fertilizers. It was beyond the scope of this paper to give an extensive overview of all advantages and drawbacks, taking into account all variables and contextual factors for all types of compost application. To compare the benefits and drawbacks of compost with other alternatives, an inclusive study must be conducted taking into account all relevant parameters. However, it is difficult to translate all the aspects of compost production, storage, transport and spreading into economic and particularly ecological values. For instance, cycle closure, carbon sequestration and the related positive effects on the emission of greenhouse gases, water holding capacity or improvement of soil structure are difficult to take into account in assessment methods such as Life Cycle Assessment (LCA) (Obersteiner and Linzner, 2007).

Composting is a biological process where microorganisms convert organic products into a stable and humus-rich product under controlled conditions, i.e., optimal conditions of moisture and oxygen to facilitate the decomposition process (Bokhorst and ter Berg, 2001; Willekens and Cloet, 2003). Unlike fast-release fertilizers such as mineral fertilizers and slurry, compost contains **large amounts of organic matter**, which enhances the SOC content (Vanden Nest et al., 2014). During three long-term field experiments in Flanders, a significantly higher SOC content was observed when farm compost (made from farmyard

residues) (D'Hose et al., 2014; Willekens et al., 2014) and VFG compost (made from vegetable, fruit and garden waste) (Tits et al., 2014) was applied. When mineral fertilizer was applied in a parallel treatment, SOC content decreased. Compost application also improves soil physical properties such as available water content (Curtis and Claassen, 2009; Weber et al., 2007) and aggregate stability (Annabi et al., 2007), which in turn protects the soil against erosion (Diacono and Montemurro, 2010). The organic carbon in compost is more **stable** and resistant to decomposition than in fresh manure or plant residues, where a larger share of the carbon decomposes after application. Compared to an equal amount of farmyard manure applied, twice as much of the applied carbon is retained in the soil when using composted farmyard manure, not taking into account the carbon losses during the composting process (Powlson et al., 2012). In addition to maintaining and improving SOC, compost is also a **source of nutrients**, which reduces the need for other fertilizers. This reduces both the cost of purchasing non-organic fertilizers and can reduce the environmental impact associated with fertilizer production and use. For instance, both D'Hose et al. (2014) and Willekens et al. (2014) observed enhanced plant available potassium contents in soil after repeated compost application. Nevens and Reheul (2003) found that silage maize needed 0 to 43 kg less mineral fertilizer ha^{-1} on plots with compost application (22.5 tonnes ha^{-1}) compared with the plots only receiving mineral fertilizer. One point of consideration is European legislation that limits the supply of nitrogen and phosphorus with the aim of preventing nutrient leaching to ground and surface water. In addition to intercropping systems and soil cover as potential means to limit nutrient leaching, a balance must be found between the supply of nutrients from carbon-rich soil improving fertilizers (for improving soil fertility), and the nutrients supplied from fast-acting fertilizers (for plant nutrition). By adding large amounts of carbon-rich materials (e.g. straw, wood chips) in the composting process, the end product enables adding a significant amount of carbon without adding large amounts of phosphorus and nitrogen. As such, the risk for nutrient leaching is lower than for composts or other organic amendments with a lower carbon-to-nitrogen or carbon-to-phosphorus ratio (Vandecasteele et al., 2014). The nutrients in compost are **released gradually** because they are already fixed in the microbial biomass (Sullivan et al., 1998), unlike the quick release from slurry, farmyard manure and nitrogen-rich crop residues. Compost application therefore helps to prevent nutrient leaching to groundwater (Grey and Henry, 1999; Li et al., 1997) and contributes to soil fertility in the long term. For example, recent research showed that long term amendments of plant-based compost did not increase phosphorus leaching as compared to amendments of dairy farmyard manure (Vanden Nest

et al., 2016). Additionally, repeated compost amendments can enhance the **biological diversity** of the soil (D'Hose et al., 2014; Steel et al., 2012) and can decrease the amount and relative abundance of plant-parasitic nematodes (D'Hose et al., 2014). This might reduce the risk of plant diseases and thus the use of pesticides and herbicides. For example, after three years of compost application, the total microbial biomass increased by 27% (Willekens et al., 2014). Moreover, pathogens and weed seeds in the feedstock mixture are suppressed by high temperatures, microbial antagonism and/or competition for nutrients, toxicity from byproducts of organic matter decomposition (e.g., ammonia, sulfides, organic acids, and phenolic compounds), and enzymatic breakdown during the composting process (Wichuk et al., 2011), in contrast to amending the soil through direct application of farmyard residues. Furthermore, a significant **reduction in volume and moisture content** is observed when composting agricultural byproducts (Bernal et al., 2009; Breitenbeck and Schellinger, 2004). This can lead to ecological and economic advantages such as more efficient transport and storage compared to the initial biomass feedstock. The more homogenous and fragmented structure furthermore results in easier spreading compared to other organic fertilizers (e.g. non-composted farmyard manure).

Besides the advantages of compost application on soil quality, compost can play an important role in the **bioeconomy** because its production does not rely on finite inputs. Compost can be produced locally on the farm, and can use biomass that is unusable in other biomass cascade[3] applications (e.g. materials not suitable for digestion or combustion) as well as byproducts from other applications (e.g. digestate remaining after bioenergy production) as feedstock. This limits the competition for biomass, as composting is a process that can be included (typically at the end) in various biomass cascades to reduce the volume and moisture content as well as stabilize and sanitize the biomass. Furthermore, compost application on agricultural soils closes organic carbon and nutrient cycles, thus contributing to sustainable soil fertility and fertilization.

[3] The biomass cascade is the mechanism for prioritizing biomass use for socially preferable products, aiming to maximize the efficient exploitation of the available biomass (De Meester et al., 2011; European Commission, 2012; Keegan et al., 2013). Although different versions of the biomass cascade rank the value of possible applications differently, the general principle in this approach is to organize the processing of biomass to produce as many applications as possible with the highest socio-economic value, until the smallest possible amount of byproducts with no further valuable use remains (De Meester et al., 2011; Keegan et al., 2013).

2.3. Materials and Methods

This study of the barriers to on-farm composting and compost use in agriculture focused on the case of Flanders, the northern region of Belgium. Flanders is characterized by a maritime temperate climate. The cultivated land surface of 622 738 ha (46%) is dominated by loamy, sandy loam and clay soils (Platteau et al., 2014). Currently, Flanders counts 24 884 farm enterprises with an average size of 25 ha. The main crops are fodder (including grassland) (56%) and arable crops (36%) mainly including cereals, potatoes and sugar beets. Flemish farms are highly specialized: 88% of all farms are specialized in either arable farming, horticulture or livestock. In 2013, 5065 ha (0.8%) was organically cultivated. In 2013, the total number of poultry, cattle and pigs was 30.8, 1.3 and 6.2 million, respectively. This intensive and often soilless animal production, combined with the limited availability of land and consequently high pressure, has led to a significant manure surplus in Flanders. In 2013 the total animal manure production represented 160.8 million kg nitrogen (an increase of 4% compared to 1991) and 26.7 million kg phosphorus (a decrease of 10% compared to 1991). After subtracting manure from import, export, processing and nutrient losses, the total amount of nitrogen and phosphorus in animal manure returned to the soil was 94.5 and 18.6 million kg, respectively (Vlaamse Landmaatschappij, 2014).

We have performed two separate studies to analyze the current compost production and use in Flemish agriculture. The first study analyzes the barriers to on-farm composting, whereas the second study investigates the farmers' beliefs about the application of compost in Flemish agriculture. In the first study, a concurrent mixed method was used; in the second, a sequential mixed method procedure was used (Creswell, 2003) (Figure 2.1). These mixed method approaches allow for both methodological and data triangulation, increasing the validity of the data (Creswell, 2003; Denzin, 2006; Johnson et al., 2007). In **the first study,** semi-structured interviews (1-3h) (referred to below as *interviews from study 1)* were conducted (November 2012 – December 2013) with 70 stakeholders from diverse stakeholder groups: farmers (15), members of diverse interest groups (e.g. NGOs and agricultural sector organizations) (13), members of research institutes and extension services (17), composting agents (9), policymakers (5) and others (11). The interviews had a series of recurring questions (Appendix Table A1), but allowed further elaboration through in-depth questions. Stakeholders were identified through snowball sampling (interviewees referred to other relevant stakeholders which in turn were interviewed (Patton, 1990)). When no new information or new names of relevant stakeholders appeared, the point of saturation

was achieved (Morse, 1991). The interviews revealed a number of barriers to on-farm composting. To further verify and reflect on the results of the interviews, 30 stakeholders participated in a focus group. Additionally, based on the revealed barriers, two questionnaires were conducted in September 2013. The first questionnaire was designed in collaboration with *Natuurpunt* (referred to below as *Q1*), the largest nature conservation association in Flanders. A population of 550 farmers who have a parcel user license with *Natuurpunt* (i.e., the farmers are mowing or have livestock grazing on certain parcels of *Natuurpunt* at no cost but subject to certain conditions set by *Natuurpunt*) received the questionnaire (response rate of 11%). The second questionnaire was conducted in collaboration with *BioForum* (referred to below as *Q2*), the Flemish organic farmers' association, and was sent to 230 organic farmers in Flanders (10% response).

The second study was part of the European CATCH-C project (EU FP7 No° 289782). A sequential mixed method approach, based on the Theory of Planned Behavior as socio-psychological framework, was used to design a questionnaire to investigate farmers' beliefs about the use of compost (Ajzen, 1988; Ajzen, 1991; Wauters and Mathijs, 2013; Wauters et al., 2010). Two regions were selected and characterized by soil texture, slope and land use. In the area surrounding Roeselare, Belgium (region of West Flanders), farmers of mixed farms (vegetables-pigs) were approached; in the province of Flemish Brabant, where loamy soils dominate the landscape, arable farmers were targeted. In both regions, semi-structured interviews (Bijttebier et al., 2015) were conducted with randomly selected farmers to reveal a list of outcomes, normative referents and control factors for the use of compost (n = 9 for mixed farms, n = 7 for arable farms). The outcomes, referents and control factors resulting from the interviews gave a first indication of what the barriers could be. However, to assess whether these are compatible with the beliefs of the population of farmers in the particular region, a large scale questionnaire was set up in each region as a second, quantitative step of the mixed method (referred to below as *Q3* for the arable farms and *Q4* for the mixed farms). Two questions were asked for each of the identified outcomes, referents and control factors (Appendix Table A2). The questionnaire was sent via mail. Sixty-one mixed farms and 121 arable farms completed the questionnaire.

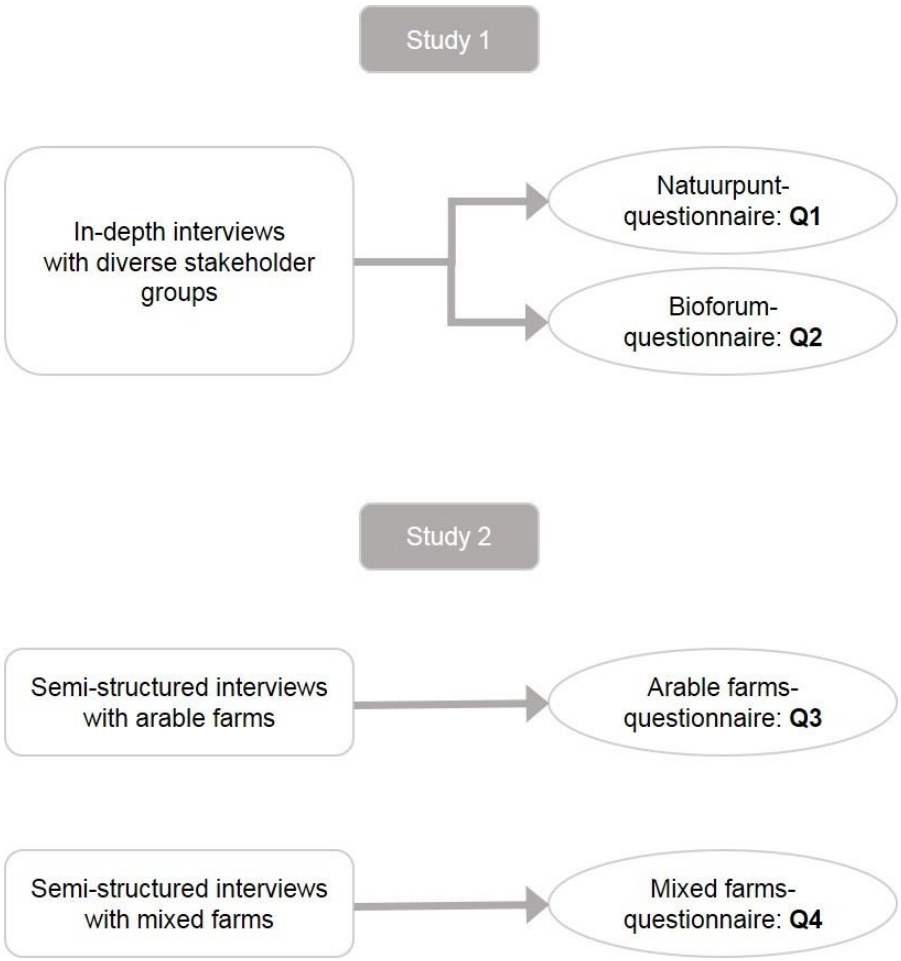

Figure 2.1 Overview of the two studies used for data collection, both based on a mixed method approach. Qualitative and quantitative research steps are represented by rectangles and ovals, respectively.

2.4. Current compost production and use of compost in Flemish agriculture

2.4.1. Compost use

Q3 and Q4 indicated that the majority of the Flemish farmers apply inorganic (mineral) fertilizers (93% of the arable farms and 97% of the mixed farms) and/or slurry (85% of the arable farms and 100% of the mixed farms) as a source of nutrients. Farmyard manure is also a widely used organic fertilizer in arable farming (adoption rate of 67%) and to a lesser extent in mixed farming (58%). Most farmers also incorporate maize grain straw (71% of the arable farms and 80% of the mixed farms). Incorporation of cereal straw is less common (24% of the arable farms and 22% of the mixed farms). The interviews of study 1 and the questionnaires confirm that only a minority of the farmers apply compost: Q3 showed that

only 8 (6.6%) of the arable farmer respondents used compost on an average of 10% of their land and only 11% of the respondents had the intention to use compost in the near future. None of the mixed farm respondents of Q4 are using compost. Q1 revealed that 11 (19%) farmers with a parcel user license with *Natuurpunt* apply compost, the majority of which (66%) are organic farmers. The higher adoption rate for compost use in organic farming was confirmed by Q2: 87% of the organic farmers who responded to the questionnaire were using compost.

2.4.2. Compost production

When farmers want to use compost in Flanders, they generally have two options: they can either produce the compost themselves (on-farm composting), or they can buy the compost from an external company that produces compost on a (semi-)industrial scale. On-farm composting is generally defined as an individual farmer recycling his own farmyard residues (e.g., crop residues, grass clippings, animal manure) into compost and using the compost on his own fields. The farmer generally piles up agricultural residues in long rows, called windrows. The interviews from study 1 indicated that only a minority of the respondents were making compost on their own farm. This was confirmed in Q3 and Q4, where none of the arable farmers using compost produced it themselves, and only 1.7% of the respondents considered on-farm composting. This is in contrast with organic farmers, where 80% of the compost users from Q2 were on-farm composters. In contrast to the small amount of agricultural byproducts processed through on-farm composting, a large proportion of biomass is composted at (semi-)industrial composting facilities. These companies are a heterogeneous group; what they have in common is that they process external feedstock, generally on a larger scale than on-farm composters. This external feedstock is either biowaste (municipal Vegetable, Fruit and Garden Waste, resulting in VFG compost) or public garden waste (resulting in green waste compost). Besides the differences in feedstock, they also differ in terms of ownership (private or public), size, composting method, compost quality, whether or not they are certified, and the target market for their compost. In 2013, a total of 40 companies converted 770 000 tonnes of biomass into 363 333 tonnes of compost (Flemish compost organization, Vlaco vzw, 2013). This corresponds to 3.0 million kg nitrogen and 0.6 million kg phosphorus, which is very low compared to the amount applied with animal manure in Flanders. The majority was sold to landscapers (garden contractors, public landscaping and private sector) (Vlaco vzw, 2013). Only 5.2% of the industrially produced compost was used in agriculture and horticulture.

2.5. Barriers to on-farm composting and compost application in Flanders

The respondents of both studies acknowledged the positive effects of compost on soil quality. For instance, the respondents in Q3 and Q4 believe that compost application has beneficial effects on aspects they believe to be important such as: improved soil fertility (Q3), more diverse soil life (Q4), better soil health (Q3), lower erosion risk (Q3), better water infiltration and drainage (Q4), increased humus content (Q3 and Q4), improved soil structure (Q4), improvement of heavy soils (Q3) and increased nitrogen mineralization potential (Q3) (Table 2.1). These results seem to indicate that most farmers recognize the biophysical advantages of compost. This suggests that the mechanisms preventing farmers from composting their farmyard residues and/or applying compost are not related to their belief in the benefits of compost use. We divided the barriers to on-farm composting and compost application into four categories: policy and institutional (P&I), market and financial (M&F), scientific and technological (S&T), and informational and behavioral (I&B) (Lee, 2014). In the explanatory paragraphs below, the appropriate category is noted between brackets.

Table 2.1 Mean belief strength (1: very untrue to 5: very true) and outcome evaluation (1: very bad to 5: very good) for the outcomes of compost application on arable farms (n = 121) (Q3) and mixed farms (n = 62) (Q4).

	Arable farms		Mixed farms	
	Belief strength	Outcome evaluation	Belief strength	Outcome evaluation
Improved long term nitrogen release by the soil	3.5	3.8		
Improved soil life	3.9	4.2	4.0	4.3
Increased humous content of soil	4.0	4.0	4.0	4.3
Lower erosion risk	3.7	4.2		
Unsure on timing of nitrogen release for crop	3.7	2.4		
More weeds	3.7	2.0		
Contains waste products	3.8	1.9		
Improved soil fertility	3.9	4.2		
Improved soil health	3.7	4.3		
Higher risk on diseases	3.3	1.9	3.2	1.6
Obtain less heavy soils	3.8	3.8		
Higher risk for excessive nitrogen residue in autumn	3.6	2.0		
Supply of nitrogen needed to digest compost	3.1	2.5		
No homogenous spread	2.5	2.2		
Better water infiltration and drainage			4.0	1.7*
Improved soil structure			4.1	4.4
Faster nutrient release compared to farmyard manure			3.1	3.6
Better soil improver than farmyard manure			2.6	4.0
More labor intensive			3.6	2.4

* This question was asked in the reverse form for the outcome evaluation: Less water infiltration and drainage.

2.5.1. Barriers to on-farm composting

Shortage of woody materials. The first legislative barrier frequently mentioned by the interviewees of the first study, is the promotion of biomass use for green energy production (EU 2020 targets) through subsidies (green certificates) [P&I1]. These subsidies make it economically more attractive to incinerate or digest biomass instead of using it in other biomass cascade applications and composting. Furthermore, the promotion of bioenergy has created a distortion of biomass (especially woody biomass) toward the energy sector. The resulting shortage of woody biomass – a crucial element in composting, as it is both a carbon source and creates the aerobic conditions during composting – has resulted in higher prices. This makes composting more expensive for farmers that do not already possess woody biomass [M&F1], thus causing an economic barrier.

Regulations for composting. For an optimal composting process and consistently high-quality compost, a farmer needs the proper combination of brown (e.g. wood chips and straw) and green (more nitrogen-rich, e.g. crop residues) feedstock. Together they provide the proper carbon-to-nitrogen ratio that the microorganisms need to break down the organic material. However, because of the highly specialized nature of farms in Flanders, both types of feedstock are generally not available at one farm, at least not in the right balance. Therefore, biomass must be purchased from a third party (external biomass) as mentioned by the interviewees of study 1. This was also illustrated by the findings that 56% of the on-farm composters of Q2 and 83% of the on-farm composters of Q1 mix material from a third party (from nature areas, other farmers or garden contractors) with farmyard residues. However, farmers are generally hindered by the required environmental license to use those external biomass sources [P&I2]. The requirements to obtain such a license, depending on the amount and type of materials processed (Table 2.2), can include a number of costly investments such as installing a concrete pad for composting, a system to capture and store run-off waste water, measurements to reduce odor, emissions and dust, and mandatory compost quality control. These strict and complex regulations impede farmers from composting, as indicated by the interviewees in study 1. This was confirmed by the non-composters of Q1, as 28% is not sure what is allowed when composting on-farm. Also, 11% of the non-composters of Q1 would compost on-farm if the regulations were less strict. When external animal manure is included as feedstock, having an environmental license is mandatory (Table 2.2). The storage of manure (and also composting of manure) on the field is forbidden from the 15th of November until the 15th of January, and is limited to maximum

one month outside that period. This implies the need for on-farm storage and composting under strict conditions, including the installation of a concrete, impermeable pad. Besides an expensive investment, this regulation demands additional farm space, creating a barrier to composting the manure under these conditions as indicated by the interviewees of study 1 [P&I3].

Table 2.2 Illustration of the complexity of Flemish regulation for on-farm composting. Adapted from *Bioforum Vlaanderen* (2013).

Vlarem I appendices		Feedstock from the own farm	Green waste from the own farm	Green waste from a third party	Manure from the own farm	Manure from a third party
Feedstock	from the own farm	No environmental licence required (*)	No environmental licence required (*)	Category 2.2.3	Category 9	Category 28.3
Green waste	from the own farm	No environmental licence required (*)	No environmental licence required (*)	Category 2.2.3	Category 9	Category 28.3
Green waste	from a third party	Category 2.2.3	Category 2.2.3	Not applicable for on-farm composting	Category 2.2.3 + category 9	Category 2.2.3 + category 28.3
Manure	from the own farm	Category 9	Category 9	Category 2.2.3 + category 9	Category 9	Category 28.3
Manure	from a third party	Category 28.3	Category 28.3	Category 2.2.3 + category 28.3	Category 28.3	Not applicable for on-farm composting

(*) The case for organic biomass from the own farm (green waste + crop residues). Use on the own fields is mandatory.

VLAREMA:
Section 1.2. Definitions > Article 1.2.1 > §2 > 33°
Green waste Compostable organic waste from gardens, parks, river banks, roadsides and nature

VLAREM I (appendices)
Category 2. Waste products > Facilities for the treatment of waste products
Category 2.2. Storage and useful application of waste products

Category 2.2.3. Storage and biological treatment of: useful application on the production site, including at home composting, and on-farm composting when only proprietary feedstock is used, and the compost is only ammend on the own parcels, is not seen as storage or treatment of waste products;

> depending on the type of waste and the capacity an environmental licence type 1, 2 or 3 is required with conditions

Category 9. Animals > Guidelinces for stables
including:
- the facilities for the treatment of animal manure that was produced on that specific site, without adding waste
- the facilities for the treatment of animal manure that was produced on that specific site with green waste from the site and the proprietary lands

> depending on the capacity, amounts, site , … an environmental licence type 1 or 2 is required with conditions

Category 28.3. > Manure treatment
Category 28.3. Facilities for the treatment of animal manure, excluding the facilities as mentioned under category 9.3 until 9.8, with a capacity of …

> depending on the capacity an environmental licence type 1 or 2 is required with conditions

Financial investment. A good quality compost relies heavily on the right balance of oxygen, water and temperature throughout the whole composting process. Monitoring and managing these parameters requires a number of specific tools and machines [M&F2]. The basic monitoring equipment required is an appropriate thermometer and a CO_2 sensor. In addition, in order to maintain the required oxygen and temperature levels, the compost needs to be aerated. Although a sufficient level of aeration can be achieved with equipment such as a front loader and manure spreader, available on many farms, a specific compost turning machine yields the best results. Such a compost turner is also often equipped with a watering device which provides the necessary water supply. Respondents of the first study indicated that a compost turner and the monitoring equipment are too expensive for an individual

farmer, consequently they are almost never available on the farm. In addition to the monitoring equipment and a turner, a pile cover should be used to avoid rainwater seeping into the compost and creating undesired anoxic conditions. Only two respondents of Q1 and 4 respondents of Q2 owned a pile cover. Furthermore, certain feedstock should be reduced in size for optimal breakdown during the composting process. The necessary shredding machines are expensive and use large amounts of fuel and are thus rarely available on the farm. Furthermore, most of the feedstock is characterized by a high volume-to-weight ratio, resulting in high transport costs, as indicated by the respondents in the interviews of study 1 [M&F3].

Time investment. In addition to the financial investment, composting also demands that the farmer makes a significant investment of time to monitor the process and turn the compost when necessary. The interviews from study 1 show that the significant time investment in these activities prevents farmers from on-farm composting, because they may compete with the farmers' primary activities [M&F4]. Moreover, due to the seasonality and fast decay of certain feedstock (e.g. grass clippings and crop residues), farmers sometimes lack the required time to compost when feedstock is available or do not have feedstock available when they can invest time into composting [M&F5]. The abovementioned financial and time investments, together with the more long-term benefits of compost application, make it difficult for on-farm composting to be profitable on the short term [M&F6].

Lack of experience and knowledge. The first study indicates that a lack of experience and knowledge might prevent farmers from composting farmyard residues [I&B1]. Moreover, the respondents mention a lack of knowledge about the regulations for on-farm composting [I&B2]. For example, almost all (96%) non-composting farmers from Q1 and one-third of the composting farmers from Q2 are not familiar with the composting regulations.

Profitability. An additional barrier revealed by the interviews of study 1 is the farmers' perception that the cost of composting followed by compost application is far higher than the cost of applying alternative fertilizers. This perception could be caused by the actual higher short-term costs, while the benefits manifests particularly on the long-term [I&B3].

2.5.2. Barriers to compost application

Complex regulations. The regulations about the use of compost in Flanders are found at the interface of several policy areas (environment, agriculture, public health) and topics (waste, manure, food safety, etc.) at different policy levels (Flemish, Belgian, European). The result

is a complex, confusing, and often contradictory legislative landscape [P&I4]. For example, on the one hand nutrient leaching has to be prevented as stated in the EU Nitrates Directive 91/676/EEC and EU Water Framework Directive 2000/60/EU, which is implemented in Flanders through the Manure Decree that restricts the total allowed number of livestock and the nitrogen and phosphorus fertilization. On the other hand, the Mid-Term Review of the Common Agricultural Policy (COM/2002/0394) states that soil quality and organic matter content must be maintained or improved and carbon must be stored (Second European Climate Change Programme). In other words, compost application is stimulated by the Mid-Term Review, while it is limited by the fertilization legislation. The legislative complexity was also confirmed by Langlais et al. (2014). Q3 revealed that the strict manure legislation makes the use of compost unattractive (Table 2.3). Some exceptions have been made for the application of compost: (i) if compost is produced on the farm from own materials and no animal manure is included, it does not have to comply with the fertilization legislation and (ii) if compost is certified, only 50% of the applied phosphorus is accounted for and 15% of the total nitrogen is accounted to be effective. However, these exceptions do not apply to compost produced on the farm using external biomass or animal manure. The interviewees of study 1 confirmed that this legislative complexity forms a barrier to compost application. Furthermore, the majority (71% of the farmers of Q1 and 100% of the farmers of Q2) believe that the legislation does not encourage the use of compost in agriculture and that these policies creates disincentives for compost use.

Manure surplus. The intensive livestock farming in Flanders creates a ready supply of slurry, which is mostly provided and spread on the field at no charge. This creates an important barrier to compost application [M&F7], as deduced from Q3 and Q4 (Table 2.3). Moreover, if farmers use manure/slurry, the use of other organic amendments such as compost is restricted due to the limited amount of nitrogen and phosphorus that may be legally applied [P&I5]. Animal farmers apply the maximum amount of nutrients from manure on their own land to limit manure surpluses and related processing costs [M&F8]. A social barrier related to this manure surplus (revealed by Q3) is that farmers often use slurry or manure from neighboring farmers as part of maintaining a good relationship [I&B4] (Table 2.4).

Availability and transport of the compost. Q3 indicates that farmers believe the local compost supply is insufficient and they are unsure about compost availability when they need it [M&F9] (Table 2.3). Furthermore, more than 0.5 tonnes of compost may not be

transported from the compost producing company to a farmer unless either a certified manure transporter or certified sender are used [P&I6]. The cost of such transport must be covered by the farmer, while slurry is transported and generally also spread at no charge (Table 2.3). Moreover, Q3 indicated difficulties in finding a transporter for the compost as well as the need for a contractor to transport and spread the compost [M&F10] (Table 2.3).

Table 2.3 Mean control belief (1: strongly disagree to 5: strongly agree) and perceived power (1: very unimportant to 5: very important) for the control factors of the application of compost on arable farms (n = 121) (Q3) and mixed farms (n = 62) (Q4).

	Arable farms		Mixed farms	
	Control belief	Perceived power	Control belief	Perceived power
Expensive transport	3.3	1.7		
More expensive compared to other organic fertilizers	3.3	1.7		
Prices are variable	3.7	2.1		
Hard to find transporter	3.5	2.2		
No appropriate machinery available for spread	2.9	2.1	3.9	2.0
Dependent on contractor to spread compost	3.9	2.5		
Not sure on availability when needed	3.5	1.9		
Low offer of compost	3.6	1.8	3.6	1.9
Slurry is spread for me, compost not	3.8	2.2		
More than enough slurry available	3.9	2.5		
Manure is easy available to me	2.5	2.1		
Not enough knowledge on composition	3.6	2.1		
No experience with compost	4.1	2.1	4.4	2.2
Much variation in quality	3.8	2.2		
I incorporate straw	2.6	2.6		
I prefer organic fertilizer of animal origin	3.8	3.1		
I prefer solid fertilizer compared to liquid	3.5	3.2		
Legislation for fertilization is too strict	4.0	2.4		
Compost is expensive			3.2	1.6
Lack of knowledge			4.0	2.0
Don't know where to get it			3.5	1.7
Vegetables do not need humus			1.5	3.4
Other alternatives to maintain humus content			4.0	2.0
Too much slurry			4.3	1.7
Humus content of my soils is good			3.2	2.4

Quality and composition of compost. The interviewees from study 1 and the arable farmers in Q3 think that the quality and price of compost varies [M&F11]. A reason for this variable compost quality can be related to the large variety in feedstock and composting circumstances [M&F12] and the often large-scale character of the compost producing companies, often causing a suboptimal composting process, as mentioned by some interviewees of study 1. To ensure sanitization, their composting process reaches high temperatures, but this can result in a biologically poor compost with low concentrations of beneficial nematodes (Steel et al., 2012). Regardless of whether the compost is actually sanitized, the respondents of Q3 and Q4 believe there is a risk for weeds and diseases (Table 2.1), making them reluctant to use compost [I&B5]. Besides the variable compost quality,

the interviews of study 1, Q1 and Q2 indicate that farmers are also uncertain about the quality and composition of the compost received [I&B6] because of the broad definition of the term 'compost' in Flanders. A reason for this uncertainty, mentioned by some interviewees of study 1, could be that the legally regulated compost types in Flanders only distinguish between feedstock, i.e., VFG compost and green compost. Once certain quality standards are met, no further differentiations are made based on quality of the compost or the target application, e.g., specific compost for agriculture, horticulture or for application in potting soils or stables [P&I7]. A last hindering factor regarding compost composition, derived from Q3, is the uncertainty about the period and amount of nitrogen release for the crop (Table 2.1) [S&T1].

Experience with compost. The respondents from Q1 and Q2, especially non-users of compost, indicate having a lack of experience with and knowledge about using compost [I&B7]. From Q3 and Q4 we conclude that the farmers' environment does not stimulate compost use: the educational system and extension services generally do not recommend compost application and farmers find only few examples from colleagues [I&B8] (Table 2.4). Farmers place the most importance on the judgment of education authorities, extension services, researchers and agricultural magazines (Table 2.4). Additionally, despite the proven long-term soil benefits of compost application, the interviewed farmers from study 1 indicate a reluctance to invest in a long-term soil improving strategy because many agricultural fields in Flanders are leased (e.g., on average 54% of the parcels in study 2 is leased) [M&F13]. Consequently, the yield of the investment might not be appropriated by the farmer when exploitation rights of the leased lands are revoked.

Table 2.4 Mean normative belief (1: very unattractive/difficult to 5: very attractive/easy) and motivation to comply (1: very untrue to 5: very true) for the referents of the use of compost on arable farms (n = 121) (Q3) and mixed farms (n = 62) (Q4).

	Arable farms		Mixed farms	
	Normative belief	Motivation to comply	Normative belief	Motivation to comply
Other arable farmers make little use of compost	3.7	3.7		
I can do animal farmers in area a favor by using their slurry/farmyard manure	3.4	3.4		
Agricultural magazines	3.2	3.8	2.7	3.5
The municipality			1.5	2.6
Extension			1.6	3.8
Producers of compost			1.6	3.3
Other farmers			1.3	3.2
Experimental results			2.6	3.7
Education			2.0	3.9

2.6. Discussion

Figure 2.2 summarizes the abovementioned barriers and their relationships. In total, 28 barriers were identified: 10 to on-farm composting, 16 to compost application and 3 barriers that affect both on-farm composting and compost application. For on-farm composting, 11 barriers, and for compost application, 13 barriers were direct barriers (continuous arrows in Figure 2.2). When only looking at the numbers of identified barriers, the figure shows that market and financial barriers are the most important category of barriers (6 for on-farm composting and 8 for compost application). For on-farm composting, policy and institutional barriers (4 in total) are the second most important, followed by informational and behavior (3). For compost application, the informational and behavioral barriers (6 in total) are the second largest hindering category, followed by policy and institutional (4) and science and technology (1) barriers. This indicates that technical aspects only play a minor role. A possible explanation is the vast amount of scientific research on composting the past decades that has resulted in a significant amount of knowledge about the composting process, its optimization and influencing factors. The number of barriers per category does not necessarily indicate the importance of a category, however; these barriers are often context- and farm-specific (e.g., farm size, type of farmyard residues and relationship with nature conservationists). Moreover, some barriers are more important because a large amount of respondents indicated these barriers as important and/or a barrier is a locus for one or more other barriers.

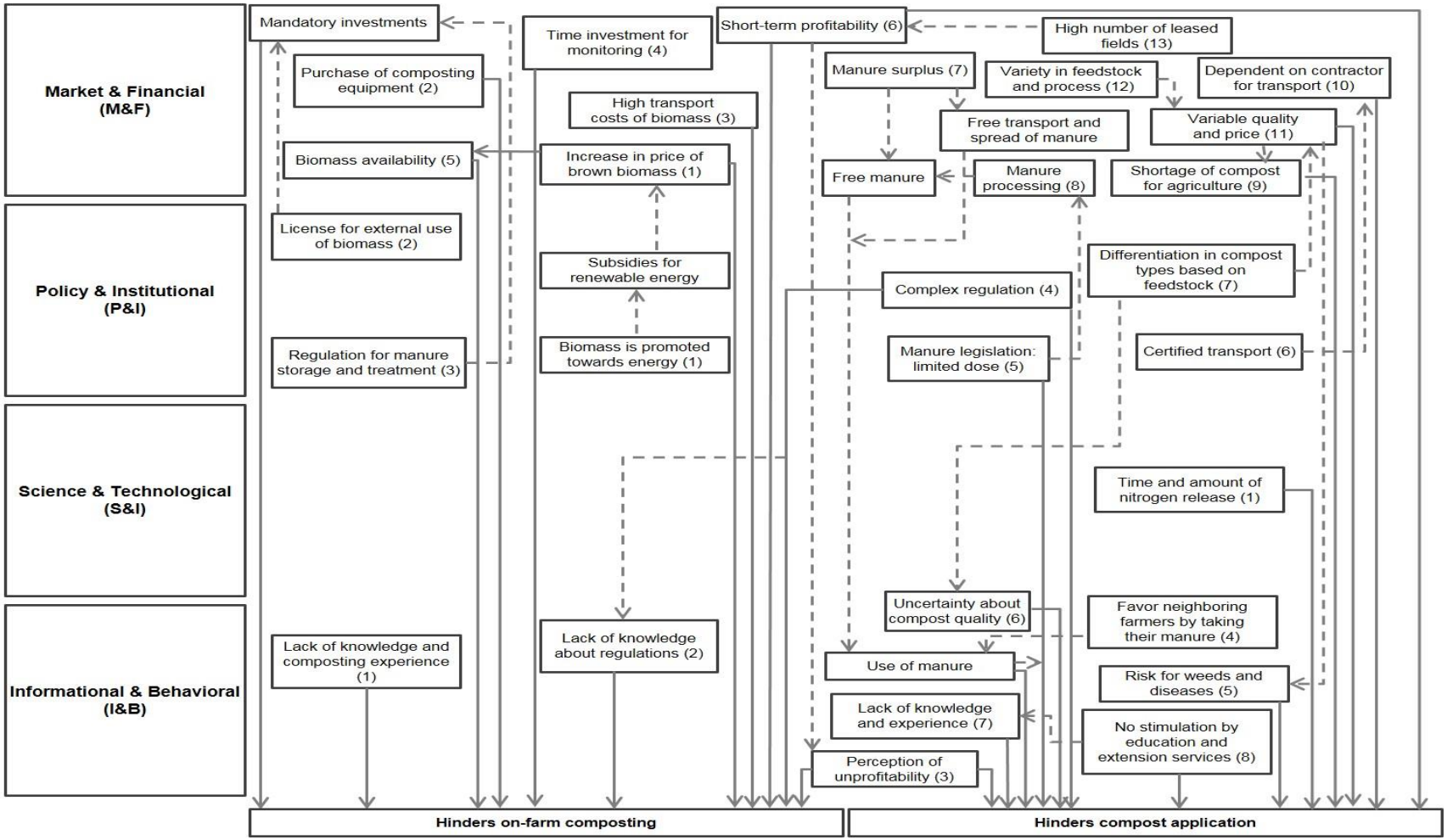

Figure 2.2 Overview of the barriers to on-farm composting and compost application in Flemish agriculture, divided into four categories: market & financial, policy & institutional, science & technology, informational & behavioral barriers. Continuous arrows are representing main direct links and dotted arrows indirect links. Per category, the number of the corresponding barrier as discussed in the text is given between brackets.

Our research shows that only a minority of the Flemish farmers currently either compost their farmyard residues and/or animal manure on their own farm or apply industrially-produced compost. Especially in conventional farming (Q1, Q3 and Q4), compost use is not common. In Q3 and Q4 none of the farmers produce compost on-farm. In organic farming, Q2 shows that 87% of the organic farmers use compost, 80% of whom produce their compost on-farm. Approximately 16% of the farmyard manure in organic farming in Flanders is composted (personal communication). Nevertheless, organic farming in Flanders only represents 0.8% of the cultivated area (Platteau et al., 2014). These high percentages could also be caused by a higher willingness of compost users to fill in the questionnaire. Another reason for the higher adoption rate of compost in organic farming might be linked to the principles of organic farming, e.g. closing nutrient and material cycles and the importance of a high soil fertility level for achieving good crop growth. The generally low adoption rates for compost use and on-farm production coincide with findings of other studies, although the adoption of composting in Flanders is the lowest of all published studies. For instance in Florida (Rahmani et al., 2004) and Illinois (Walker et al., 2006) 19% and 25% of the respondents produced their own compost, respectively. Walker et al. (2006) further showed that the majority (64%) was not interested in on-farm composting. Data on on-farm compost production in Europe are currently lacking, in contrast to figures on compost use. Figures from the European Compost Network (Barth et al,. 2008) from European regions with levels of livestock farming similar to Flanders show that industrially produced compost is used more often in agriculture and horticulture compared to Flanders: 60%, 57% and 43% of the produced compost was used by farmers for the Netherlands, Denmark and the UK, respectively. Possible explanations are more flexible legislation (Barth et al., 2008), a larger share of organic farming, more extensive livestock farming (less animals per ha, less manure excess), more stimulation for compost use by referents (Pronk et al., 2014), and a higher awareness of soil quality.

Although only a very small minority of farmers apply compost, the majority of the interviewed and surveyed farmers are convinced that compost application has positive effects on soil fertility. A study by Danso et al. (2006) showed similar results: all compost users and 80% of the non-users were convinced of the advantages of compost as a soil improver. In agreement with these results, farmers in the study of Walker et al. (2006) mentioned better soil characteristics, crop health, germination and weed control, and a better resistance to diseases, drought and run-off as beneficial aspects of compost application.

The reasons for the discrepancy between the recognition of the added value of compost and the actual use are manifold and diverse, as our results illustrate. All categories of barriers, except for technological barriers, cover a large number of barriers (Figure 2.2). Tey et al. (2014) looked at factors influencing the adoption of sustainable agricultural practices in Malaysia and also found that no single dimension can (best) explain adoption.

Based on the results of this study, we assume that for many farmers, the most important barriers are market and financial in nature. Warburton and Sarfo-Mesah (1998) also observed this in their study on concerns and constraints in the application of compost from urban organic waste. They state that constraints are more economic in nature than technical or cultural. In our study, transport costs were a market and financial hindering factor in three ways: (i) the high cost of transporting biomass, (ii) the dependency on a licensed contractor or sender and (iii) the competition with the transport of manure at no cost. Other studies have also identified high transport costs for compost as problematic. Tey et al. (2014) found that compost use in Malaysian agriculture was limited by transport costs, as the viable transport range, even with subsidies, was limited to 35 km. A study by Maxwell and Zziwa (1992) also mentions transport cost as a significant barrier to compost application.

One of the most important market and financial barriers is the manure surplus in Flanders. As a result of this surplus, livestock farmers spread their excess manure on the fields of neighboring farmers at no cost. Furthermore, the Manure Decree constrains the application of other organic amendments such as compost. Also in other countries, low-cost, abundantly available manure is perceived to be a hindrance for compost application (Danso et al., 2006).

The cluster of uncertainty and variability in compost quality and price and the shortage of compost for agricultural purposes were also mentioned as barriers in other countries (e.g. Rahmani et al., 2004; Walker et al., 2006). Farmers that lease parcels are hesitant to use compost on the leased parcels due to the long-term nature of the investment on a parcel that they may no longer be farming in a few years. This ownership issue has also been put forward as a barrier to energy efficient investments in construction; owners are reluctant to invest because their tenants will enjoy the cost reduction instead of the owner (Faber and Hoppe, 2013). Danso et al. (2006) also observed this phenomenon; Ghanaian farmers did not want to invest in long-term soil amelioration mainly because of the insecurity of land tenure.

Within the informational and behavioral category, a lack of knowledge and expertise hinders farmers from both using compost and making it themselves. Farmers indicate a lack of

knowledge and experience about how to make compost and how to best apply it (e.g. appropriate application dose and timing). This lack of knowledge was also mentioned as a barrier in the study of Walker et al. (2006) where the farmers indicate a lack of knowledge about the economic advantages and disadvantages in producing compost. Furthermore, a lack of knowledge about the complex regulation related to compost production and application was also frequently mentioned as a hindrance by the farmers.

Besides the complex legislation regulating compost, several other policy and institutional barriers seem to be linked to barriers in other categories (Figure 2.2). For instance, the policy that stimulates the production of green energy seems to cause an increase in (brown) biomass prices. Also, the regulations related to manure storage and treatment and the licensing regulation for the use of external biomass are contributing to the high compost production costs. Consequently, as many of the policy barriers tend to contribute to or create other barriers, this category might be considered to be almost as important as the market and financial category in explaining why only a limited number of Flemish farmers produce and apply compost.

Lastly, it should be noted that other adaptations in the agrosystem (e.g., changes in crop rotation, fertilization strategy, stable system) might increase the effectiveness and the on-farm use of compost. However, it was beyond the scope of this paper to analyze those systemic issues and the related barriers.

2.7. Recommendations

Based on the abovementioned insights, five types of recommendations are suggested that could potentially alleviate a number of the identified barriers. First, we suggest exploring **alternative forms of compost production**. Currently, compost is almost exclusively produced by either a professional, (semi-)industrial composter or produced on-farm by the farmer. One potential alternative could be outsourcing the on-farm production process to a service provider. These service organizations would be responsible for the setup and monitoring of the on-farm composting process. This type of production would address a number of barriers: the farmer would have to invest significantly less time in producing the compost and would not need to purchase the required equipment, i.e. the compost turner and monitoring equipment. Additionally, this would circumvent the informational hindering factor of lack of knowledge about the composting process and experience in on-farm

composting. Another alternative means of production might be a collaboration between different farmers and/or nature conservationists and stakeholders involved in landscape management. In that way, biomass that is not suitable for an industrial composting company (too seasonal, too small amounts, long transport distances) could be centralized in a 'local biomass hub' where the biomass of the different partners could be composted. Organizing production in this way might also reduce the time investment and the production and transport costs for the individual farmer. Additionally, the cost of purchasing the required composting equipment, the mandatory investments and the required tasks could be shared among participants. Moreover, with the involvement of the right partners, an increasing amount of complementary (green/brown) biomass will be available to the cooperative. Similar cooperatives for sharing composting equipment already exist, e.g., in France (CUMA, 2015) and Germany (Maschinenring, 2015).

A second type of recommendation is some form of **financial stimulus** for on-farm composting to compensate for the high production cost associated with compost. This financial stimulus could be a direct subsidy for compost production. Alternatively, the government could issue carbon certificates to farmers who return carbon to the soil by using organic fertilizers such as compost. These financial stimuli would primarily ease the investment cost, but could also help to trigger farmers to apply compost on leased land.

Third, we recommend searching for **alternatives** to certain types of biomass, especially the **brown woody biomass** being increasingly used for the production of bioenergy. Examples of such alternatives that are less suitable for combustion but are potentially interesting for composting, are woody byproducts that result from pruning or harvesting shrubs and trees (available to fruit and tree growers), or from the management of landscape features. Currently, these woody byproducts are shredded and applied to the soil, or sometimes burned in the case of fruit trees. The landscape or nature conservation management sector in Flanders produces biomass such as structure-rich grass clippings that cannot be used as fodder or litter, residual wood from forests or landscape management, sod cuttings or chopped material from heathland management, reed from bank management or litter from forest conversion. All of those could be used as a source of carbon and could provide structure in the compost pile; however, managing the logistics of these often geographically dispersed, small amounts of biomass in a cost-efficient manner, without negative environmental impacts represents a challenge (EU Forest Strategy).

Fourth, a **certain degree of flexibility in current policies and institutional arrangements** could also stimulate compost production and application. Related to the brown biomass availability, we propose a policy alteration to level the playing field for all types of biomass valorization, including composting, instead of the current policy that favors energy production. Another policy recommendation is to simplify the regulations for on-farm production of compost where possible. For example, research (e.g. Ulen, 1993) seems to indicate that the mandatory concrete pad may not be necessary if the right precautions are taken (e.g., using a pile cover, using enough structure-rich material, regularly changing the location of the compost pile, etc.). However, further research is necessary to confirm those results. Another example is the license currently required when transporting and using external sources of biomass. Certain types of biomass either could be no longer categorized as waste, or an exception to the licensing requirement could be incorporated. In Flanders, a specific material with a specific application can be categorized as 'biomass' instead of 'waste' via a declaration for raw materials. If groups such as nature conservation services would request this declaration for grass clippings to compost at a local farm, the grass clippings are no longer 'waste', thus this would warrant an exception to the licensing requirement. One example of such an exception can be found in the Netherlands, where grass clippings from nature conservation areas (external biomass) can be used in on-farm composting without extra regulations when the clippings are transported over a limited distance (Environmental Management Act, Chapter 10). Abolishing or adjusting some of the regulations would help alleviate hindering investments and regulatory complexity. In addition to more flexible regulation, the now fractured legislation could be brought together into one 'compost code'. The rules and regulations for compost use and applications are now scattered over several legislative codes (e.g., the Manure Decree, VLAREM: Flemish Environmental Permitting Regulations, VLAREMA: Flemish regulation concerning the sustainable management of materials and waste, Federal fertilizer regulation) and legislative levels (European, National, Regional), causing confusion and complexity. The last policy and institutional issue would be to create a legal division of compost types dependent on the application (end-user) rather than the current division related to the feedstock used (VFG or green compost). Moreover, further division by certification labels with other and/or additional parameters can be created for compost to be used for landscaping and agriculture. The Netherlands have been experimenting with different labels since 2014. They have compost labels to differentiate between compost for agriculture, horticulture and animal husbandry (Keurcompost, 2014). This might help end the current situation of several diverse

compost quality standards proposed by several official and private organizations (Bernal et al., 2009). As a result, farmers will have a better idea of the offered compost quality and its characteristics, which could potentially reduce the variability in price and quality per compost type.

The fifth and final recommendation is to address the farmer's lack of knowledge and experience with compost. The recommended alternative production forms and increased flexibility and/or simplification of rules might already be steps toward alleviating the lack of knowledge and experience about composting and about the regulations. However, **more and better dissemination of information** for farmers that addresses the different aspects of composting, i.e., production, application (matching needs of cropping systems with compost quantities and specifications) and regulation, would reduce or eliminate a large number of informational barriers. Because farmers are sensitive to information from education and extension services, such services should organize activities and awareness programs to increase farmers' familiarity with compost and on-farm composting. In this way, the education and extension services would promote composting to farmers more often than they currently do.

2.8. Conclusions

Research on what hinders farmers from composting on their farm and applying compost on their fields is scarce. This paper is the first to give a general overview of the different barriers to on-farm composting and compost application in Flemish agriculture. We found a relatively large number of diverse (financial, informational and legislative) barriers. More specifically, the shortage of woody biomass, strict regulation, considerable financial and time investment, and lack of experience and knowledge are hindering on-farm composting. The complex regulation, manure surplus, variable availability and transport of compost and variable compost quality and composition are barriers to apply compost. This paper also offers a number of suggestions and recommendations that could potentially remedy a number of the identified barriers. More research and evaluation should be performed to gain more insights into the viability of the recommendations and their impact on the agricultural sector. Furthermore, gathering data on on-farm compost production in neighboring countries and comparing the barriers identified in this paper with those in neighboring countries could yield valuable further insights.

CHAPTER 3

Ensiling, composting and anaerobic digestion of vegetable crop residues

After: Viaene, J., Agneessens, L., Capito, C., Ameloot, N., Reubens, B., Willekens, K., Vandecasteele, B., De Neve, S., 2016. Co-ensiling, co-composting and anaerobic co-digestion of vegetable crop residues: Product stability and effect on soil carbon and nitrogen dynamics. Manuscript submitted for publication.

Abstract

N-rich vegetable crop residues pose a high risk for N losses during autumn. Removal and conservation of these residues followed by reuse in the field could contribute to improved recycling of nutrients, but some form of processing is required to allow storage before re-application. We have compared co-ensiling, co-composting and anaerobic co-digestion as conservation and valorization options for fresh crop residues. We studied (1) the product quality and stability and (2) the short-term effects of application of these silages, composts and digestates on soil C and N mineralization and N_2O emissions. Ensiling resulted in highly biodegradable products with a low pH (4.2-5.2) and more NH_4^+-N compared to composts. Consequently, soil incorporation of silages resulted in higher net C mineralization (up to 47% after 82 days) and microbial biomass C (up to 93 µg C g^{-1} soil after six weeks), and temporary N immobilization (up to 42 mg kg^{-1} soil). Digestates and composts led to lower C mineralization rates (between 2 and 27%) and microbial biomass C (max. 51 µg C g^{-1} soil) and no net N immobilization nor mineralization. Application of digestates resulted in high mineral N contents (47-192 mg kg^{-1} soil) and a decrease of the soil pH. In all three treatments, short-term N_2O losses after soil application were very small (< 0.11 kg N ha^{-1} after 12 days). Growers can choose the most appropriate treatment option and application moment and location, depending on the local soil and crop requirements and the on-farm facilities. Furthermore, we conclude that the parameters 'biodegradation potential' (based on the biochemical composition) and 'oxygen uptake rate' have potential as less time-consuming proxies for C mineralization to assess the product stability.

3.1. Introduction

After harvest in autumn, vegetable crop residues left on the field may cause nitrate leaching because these residues are often characterized by large amounts of biomass with a high nitrogen (N) content and low C/N ratio (Chaves et al., 2007). Additionally, N mineralization and nitrification rates are still relatively high in autumn (De Neve and Hofman, 1996). For these reasons, removal of these residues from the field is usually recommended. Preserving and valorizing removed field residues and crop residues generated off-field (e.g., from on-farm leek cleaning) is quite a challenge given their tendency to decay quickly. Possible conservation and valorization options for these N-rich crop residues include co-composting, anaerobic co-digestion (AD) and co-ensiling with drier bulking agents more rich in C. Reuse of crop residues for producing organic fertilizers and soil improvers as composts, digestates and silages, closes nutrient cycles locally and improves or maintains soil quality.

Co-composting of crop residues stabilizes and sanitizes organic material, which generates a valuable soil improver and slow-release fertilizer. The effect of compost amendment on soil dynamics is well documented: generally the plant-available N and C mineralization are low, which decreases the risk for leaching to soil and surface waters (Amlinger et al., 2003) and helps to increase topsoil organic C (D'Hose et al., 2016). AD of manure with agricultural wastes is a technology for producing biogas as a renewable energy source (Möller, 2015). The remaining digestate can be reused as a fertilizer. More research is needed on the influence of feedstock on digestate composition and the effects of digestate application on soil dynamics (Möller, 2015). Furthermore, to the best of our knowledge, there are no studies on the effect of co-digesting of silages based on vegetable crop residues. Recent research has shown that co-ensiling of vegetable crop residues with maize straw is a simple and low-cost strategy to preserve vegetable crop residues over winter (Agneessens et al., 2015). The co-ensiled crop residues can then be used either directly as fertilizer, as feedstock in composting and biogas plants or as a feed supplement for livestock (Agneessens et al., 2014).

This paper compares co-ensiling of fresh crop residues as an alternative conservation and valorization option with (1) fresh crop residues (considered here as *negative reference*), and with (2) co-composting and (3) AD of crop residues after field removal (both considered here as *positive references*), thereby producing fertilizers and soil improvers. Composts and digestates were included as positive reference materials, as more is known about their stability and effects on soil N and C dynamics, they are characterized by stable OM and

show little C and N mineralization after soil application. Therefore, we hypothesized that (i) silages would be more readily degradable than composts and digestates; and (ii) silage amendment would result in N immobilization, higher C mineralization and higher N_2O emissions compared to compost and digestate application. To test the first hypothesis, we compared different stability parameters, i.e., C/N ratio, biodegradation potential, oxygen uptake rate (OUR) and C mineralization. To test the second hypothesis, N and C mineralization and N_2O emissions were monitored during lab incubations over a period of several months.

3.2. Materials and Methods

3.2.1. Description of co-ensiling, anaerobic co-digestion and co-composting trials

This study included two trials. In trial 1, crop residues from leek (*Allium porrum* L.) and white cabbage (*Brassica oleracea convar. capitata var. Alba*) were collected in fall 2012. Residues of white cabbage were mechanically harvested with a tractor-pulled Peruzzo1600 flail cutter. Leek residues resulting from cleaning and preparing leek for the fresh market were collected on-farm. In trial 2 similar fresh leek residues (*FL*) were collected in January 2014. Co-composting was executed at field scale, while co-ensiling and anaerobic digestion were simulated at lab scale. Composting and ensiling were conducted at the Institute of Agricultural and Fisheries Research (ILVO) in Merelbeke, Belgium, while AD was executed in a lab-scale batch test at Inagro, Rumbeke-Beitem, Belgium.

3.2.1.1. Co-ensiling and anaerobic co-digestion processes

The vegetable crop residues were cut by hand (5 cm x 5 cm) and mixed in a 1:1 (trial 1) and 60:40 volume ratio (trial 2) with chopped maize straw (< 1 cm, mechanically chopped stems and leaves only). The mixtures were ensilaged in silage buckets of 15 L (Agriton, Mesen, Belgium) in four replicates per treatment. The buckets were sealed (oxygen-free) and contained a reservoir to collect possible leachate from the silage. The silages with cabbage and leek from trial 1 are referred to below as *CS1* and *LS1*, respectively, and the silage with leek from trial 2 as *LS2*. Afterward ensiling, the silages were anaerobically digested in two parallel batch tests, after mixing them with inoculum (digestate that was incubated for 1 week at 38°C) to simulate the environment of a digester. The digestates with silage of

cabbage and leek from trial 1 are referred to below as *CD1* and *LD1*, respectively, and the digestate with silage of leek from trial 2 as *LD2*.

3.2.1.2. Co-composting processes

The leek and cabbage residues were co-composted on a concrete pad in a windrow composting system. To ensure a good composting process, vegetable crop residues (high moisture and N content), should be co-composted with C-rich bulking agents to decrease the moisture content, increase the C/N ratio and minimize N losses during composting (Nolan et al., 2011). For trial 1, the C-rich bulking agents consisted of a mixture of wheat straw (1% v/v), maize straw (21% v/v), poplar bark (16% v/v) and wood chips (19% v/v), which was mixed with cabbage residues (*CC1)* on November 26th 2012 and with leek residues (*LC1)* on December 5th 2012 (each 43% v/v). In the second trial, on January 16th 2014, the leek residues (17% v/v) were mixed with maize straw (28% v/v), wood bark (11% v/v) and either chopped heath biomass (*LC2*$_{heath}$) or strawberry substrate (*LC2*$_{strawberry}$) (44% v/v). Chopped heath biomass (i.e., biomass from heathland management) or spent strawberry substrate (i.e., growing medium at the end of the growing season of strawberry culture), were tested as an inexpensive alternative for wood chips. The feedstock materials were combined to obtain feedstock mixtures with a similar C/N ratio per trial (C/N around 43 in trial 1 and 30 in trial 2). The piles were turned using a compost turner (TG 301, Gujer Innotec AG, Switzerland) to maintain aerobic conditions and ensure optimum moisture content. LC1 and LC2 were turned eight times, while LC2$_{heath}$ and LC2$_{strawberry}$ were turned 10 and 12 times, respectively. The compost piles were covered with a gas-permeable geotextile (TopTex) to protect from them precipitation. The piles were 12 m long, 3 m wide and 1.5 m high. Temperature (Digital Thermometer GTH1150) and CO_2 (Brigon Messtechnik D-63110 Rodgau) were monitored as the average of four point measurements along the length of the piles. In trial 1, pile temperatures reached a maximum (> 65°C) shortly after the composting process started (Appendix Figure A1). The CO_2 concentrations peaked at the beginning of the process and were under the detection limit after two weeks. In trial 2, pile temperatures of LC2$_{heath}$ reached a maximum of 57°C after eight days, whereas in LC2$_{strawberry}$ temperatures remained ≤ 42°C (Appendix Figure A1). Therefore, extra fresh leek residues were added to both piles on day 19 to stimulate the microbial activity and increase temperatures. Because the CO_2 concentrations in both piles, especially in LC2$_{strawberry}$, were higher than 20% in the beginning of the composting, these piles were turned frequently. Because CO_2

concentrations remained high, an extra bulking agent (poplar bark) was added on day 28. Compost trials were ended after 19 weeks for trial 1 and 11 weeks for trial 2.

3.2.2. Characterization of feedstock and composts, digestates and silages

The feedstock materials and mixtures, as well as the end products after composting, AD and ensiling were sampled in four replicates. Compost and silage samples were analyzed for physico-chemical parameters: fresh bulk density and dry matter (DM) content at 105°C (EN 13040), organic matter (OM) and ash content by mass loss on ignition at 550°C (EN 13039), pH-H_2O (EN 13037), electrical conductivity (EC) (EN 13038) and total N content (Dumas EN 13654-2). Total Ca, K, Mg and Na concentrations were measured by a charge-coupled device (CCD) simultaneous with Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES) (VISTA-PRO, Varian, Palo Alto, CA), after ashing and digestion with 7N HNO_3. Total P was measured in the same extract with a Varian CARY 50 Spectrophotometer. Water soluble NO_3-N was measured after extraction (1:5 v/v soil water extraction ratio) according to EN 13652 with a Dionex DX-600 IC ion chromatograph (Dionex, Sunnyvale, CA). Water soluble NH_4-N was measured with a Skalar SAN++ flow analyzer. As an indicator of stability, oxygen uptake rate (OUR) from the microbial oxygen consumption was measured from 20 g product (< 1 cm fraction) in 200 mL buffered nutrient solution (with N-allylthiourea (6.25 mg per flask) as nitrification inhibitor and a phosphate buffer to buffer the suspension at pH 6.5) in a 1 L Schott flask during five days of shaking at 120 rpm in a closed OxiTop respirometer at 20°C, based on the method reported in Grigatti et al. (2011). The pressure course in the bottle headspace was continuously recorded using the OxiTop device (WTW GmbH, Germany), in which the CO_2 was trapped by Sodalime (Merck). OUR was expressed as mmol kg^{-1} OM h^{-1}. Further, the cell wall components were measured according to the method of Van Soest et al. (1991). The biodegradation potential can then be estimated by the (hemicellulose + cellulose)/lignin ratio with hemicellulose = neutral detergent fiber (NDF) - acid detergent fiber (ADF), lignin = acid detergent lignin (ADL) and cellulose = ADF - ADL (Van Soest et al., 1991). The cell wall components were expressed as a percentage of OM content. The decrease in biodegradation potential of the feedstock mixtures can be used to monitor the degree of stabilization during the processes.

Feedstock materials and mixtures before composting were analyzed for fresh bulk density, DM, OM, total N and P content and biodegradation potential (only for the mixtures). Digestates were analyzed for DM, OM, total N content, OUR and biodegradation potential as described above. Pesticide residues on $LC2_{strawberry}$ were analyzed at Primoris Belgium,

according to the standard procedures (GC/MS/MS and LC/MS/MS, QuEChERS method described by Payá et al. (2007)).

3.2.3. N and C mineralization and N_2O emissions with composts, digestates and silages

C and N mineralization (trial 1 and 2) and N_2O emissions (trial 2) were assessed in laboratory incubations for soil amended with composts, digestates and silages. Two different soils (both a sandy loam (USDA texture triangle) soil from the 0-30 cm layer, air-dried and sieved < 2 mm fraction) were used for trials 1 and 2. The initial soil characteristics in trials 1 and 2 were pH-KCl = 5.55 and 6.23, respectively; TOC = 0.95 and 0.69%, respectively; hot water extractable C = 878 and 1163 mg kg^{-1} DM, respectively; NH_4^+-N = 5.8 and 1.06 mg kg^{-1} DM, respectively; NO_3^--N = 2.2 and 6.09 mg kg^{-1} DM, respectively; P-$CaCl_2$= 5.9 and 4.0 mg kg^{-1} DM, respectively and P-AL = 660 and 226 mg kg^{-1} DM, respectively.

The ensilaged crop residues were cut into small pieces of ca. 2 cm^2 and the composts were sieved over a 10 mm mesh prior to soil application. Control treatments (*CON1* in trial 1 and *CON2* in trial 2), i.e. without organic amendments, were included to allow a comparison between both trials. Each treatment had three replicates. The products from trial 1 were added on a N (104 kg N ha^{-1}) base equivalent to the amount of cabbage residue that was harvested from the field (47.5 tonnes fresh weight ha^{-1}). Similarly, the products from trial 2 were added on a N (69.2 kg N ha^{-1}) base equivalent to the reapplication of 20 tonnes ha^{-1} fresh leek that was removed from the field. For the silages, a fixed amount (60 kg N ha^{-1}) of additional mineral N fertilizer was added as NH_4NO_3 (35% N, Merck), because N immobilization was expected based on previous research (Agneessens et al., 2015) (treatments $CON1_{60N}$, $CS1_{60N}$, $LS1_{60N}$, $CON2_{60N}$ and $LS2_{60N}$). An overview and description of all trials and C and N loading rates of each treatment is presented in Table 3.1.

Table 3.1 Overview of the treatments in the N mineralization (N_{inc}), C mineralization (C_{inc}) and N_2O experiments and the respective FM, OM and N input through fresh and processed crop residues and N fertilizer. FM: fresh matter, OM: organic matter, $N_{total,product}$ = the total amount of N added with the crop residues.

		Processed crop residues			Conducted experiment		
Treatment	Description	FM	OM	$N_{total,\ product}$	N_{inc}	C_{inc}	N_2O
		t ha^{-1}	t ha^{-1}	kg ha^{-1}			
CON1	Control soil (0N)	0	0	0	x	x	
$CON1_{60N}$	Control soil + 60N	0	0	0	x		
CC1	CON + composted cabbage	31.9	4.1	104	x	x	
LC1	CON + composted leek	33.0	4.7	104	x	x	
CS1	CON + ensilaged cabbage	47.5	3.7	104		x	
LS1	CON + ensilaged leek	47.6	5.7	104		x	
$CS1_{60N}$	CON + ensilaged cabbage + 60N	47.5	3.7	104	x		
$LS1_{60N}$	CON + ensilaged leek + 60N	47.6	5.7	104	x		
CD1	CON + digested cabbage	25.3	3.7	104	x	x	
LD1	CON + digested leek	25.3	3.0	104	x	x	
CON2	Control soil (0N)	0	0	0	x	x	x
$CON2_{60N}$	Control soil + 60N	0	0	0	x		x
FL	CON + fresh leek	27.4	1.5	69	x	x	x
$LC2_{heath}$	CON + composted leek/heath chopper	17.8	2.7	69	x	x	x
$LC2_{strawberry}$	CON + composted leek/strawberry substrate	21.6	3.5	69	x	x	x
LS2	CON + ensilaged leek	26.8	2.5	69		x	x
$LS2_{60N}$	CON + ensilaged leek + 60N	26.8	2.5	69	x		x
LD2	CON + digested leek	55.2	1.7	69	x	x	x

3.2.4. Nitrogen mineralization and microbial biomass C

Soil was thoroughly mixed with either ensilaged, composted or digested crop residues and demineralized water and subsequently placed in PVC tubes (h = 12 cm, r = 2.3 cm). The mixture was then gently manually compacted to obtain a bulk density of 1.4 g cm^{-3} and a moisture content of 50% water filled pore space (WFPS). The tubes were covered with a single layer of gas permeable Parafilm® to avoid water evaporation and subsequently incubated at 15°C and 70% relative humidity. The tubes were weighed frequently and demineralized water was added as needed to maintain a WFPS of 50%. Soil mineral N (N_{min}) (ISO 14256-2), pH-KCl (ISO 10390) and DM content were analyzed at the start of the incubation and every 2-2.5 weeks for 113 days. The net N mineralization ($N_{min,net}$) on each sampling date was calculated as the difference between $N_{min,product}$ and $N_{min,control,}$ with $N_{min,product}$ the amount of mineral N released in the soil amended with composted, ensilaged

or digested crop residue with or without N fertilizer and $N_{min,control}$ the amount of N released in the unamended soil.

After 6 and 12 weeks of incubation the soil microbial biomass C (MBC) was measured by fumigating the soil mixtures with chloroform for 24h in the dark and subsequent extraction with 0.5 M K_2SO_4 (1:2 fresh soil to extraction solvent ratio). The C contents of the fumigated and non-fumigated extracts were determined with a TOC analyzer (TOC-VCPN, Shimadzu Corporation, Kyoto, Japan). Microbial C was calculated as the difference in extractable C fraction of the fumigated and non-fumigated samples.

3.2.5. Carbon mineralization

Soil was mixed with ensilaged, composted or digested crop residues similarly to the N mineralization experiment (see section 3.2.4 above) to obtain a bulk density of 1.4 g cm^{-3} and a fixed moisture content of 50% WFPS. The moisture content was determined gravimetrically throughout the experiment and demineralized water was added as needed to maintain 50% WFPS. The mixtures were placed in air-tight glass jars of 2 L together with a 15 mL 1M NaOH vial to trap the emitted CO_2. The CO_2 emitted during the incubation was subsequently measured by periodical removal of the NaOH vials and titration with 1 M HCl in the presence of $BaCl_2$ (Vance et al., 1987). After removal of the vials, the glass jars were left open for 3h to replenish the soil atmosphere with oxygen. The soil was incubated at 15°C for 115 (trial 1) and 82 days (trial 2).

A parallel first- and zero-order model, dividing the added OM in an easily decomposable and a more stable pool, gave the best fit to the C mineralization data of all treatments, except for LC1, CC1, $LC2_{heath}$ and $LC2_{strawberry}$ (Sleutel et al., 2005):

$$C_{min,net}(t) = C_{af} * \left(1 - e^{-k_f * t}\right) + k_s * t$$

With $C_{min,\ net}$ (t) the net C mineralization (mg CO_2-C kg^{-1} dry soil) at time *t* (days), C_{af} the easily decomposable C pool (mg C kg^{-1} dry soil) , k_f the mineralization rate of the easily decomposable C pool and k_s the mineralization rate of the more recalcitrant C pool (mg CO_2-C mg^{-1} C day-1).

The C mineralization data of LC1, CC1, $LC2_{heath}$ and $LC2_{strawberry}$ could be described by a linear model:

$$C_{min,net}(t) = C_{min,0} + k * t$$

where the intercept $C_{min,0}$ is the C content in the soil amended with composted, ensilaged or digested crop residue at time t = 0 and *k* is the mineralization rate.

The relative net C mineralization ($C_{min,net}$ in %) from the ensilaged, composted or digested crop residues on each sampling date was calculated as follows:

$$C_{min,net} = \frac{C_{min,treatment} - C_{min,control}}{C_{treatment,total}}$$

with $C_{min,\ treatment}$ and $C_{min,\ control}$ the amount of CO_2-C evolved from the treatments with added composted, ensilaged or digested crop residues and the control treatment, respectively. The total amount of organic C added with the composted, ensilaged or digested crop residue is denoted as $C_{treatment,total}$.

3.2.6. N_2O emissions

Before the start of this experiment, soil was pre-incubated at 10% (w/w) moisture content for 7 days to avoid a sudden peak of microbial activity at the start of the actual experiment (Moeskops et al., 2010). After pre-incubation, soil was thoroughly mixed with organic amendments and filled in PVC columns (r = 2.4 cm) and compacted to reach a bulk density of 1.27 g cm^{-3}. Demineralized water was added to reach a moisture content equivalent to 75% WFPS. The samples were put in airtight closed glass jars with a lid and septum and incubated in a dark room (19°C). The first gas samples were taken from the headspace 2 h after closing the jars. After gas sampling, the jars were left open for 0.5 h to circulate ambient air inside. Soil moisture loss was replenished by adding distilled water when moisture content dropped by more than 0.60 g. Gas was sampled twice per day for 12.5 days. For N_2O measurements 12 mL glass exetainers® (Labco Limited, Ceredigon, UK) were pre-evacuated three times consecutively using a vacuum pump and flushing with He. Finally, a vacuum of 0.04 mbar was established in the exetainers. Headspace gas samples were collected from the closed containers using an air-tight syringe (venoject multi-sample needle, Terumo, Heverlee, Belgium) and were stored in the pre-evacuated exetainers until measurement. The N_2O concentration of the headspace gas samples was measured by manual injection into a Thermo Electron Trace GC Ultra gas chromatograph equipped with an electron capture detector (ECD) (Interscience, Breda, The Netherlands; injector: 120°C; detector: 350°C; oven: 40°C; carrier gas and flow rate: N_2, 30.7 mL min^{-1}; column type and length: Hayesep Q Packed column, 2 m). Calibration curves for each treatment were

obtained before each measurement by injecting 200, 400, 600, 800 and 1000 µL of a N_2O standard gas (23 ± 1.5 µL N_2O-N L^{-1} He).

3.2.7. Statistical analysis

A one-way ANOVA and post-hoc Scheffé test were used ($p < 0.05$) to assess the significant differences in product quality between the treatments and treatment effects on MBC and cumulative N_2O emissions. Dunnett's two-tailed *t*-test was used to compare the soil pH after addition of fresh and processed crop residues with the control soil. Statistical tests were carried out using using IBM SPSS Statistics for Windows, Version 21.0 (IBM Corp., Armonk, NY, USA). A parallel first- and zero-order kinetics or linear model was fitted to the C and N mineralization data using the open-source software platform R (version 3.1.3; R CoreTeam, 2015).

3.3. Results

3.3.1. Characterization of feedstock, composts, digestates and silages

The characteristics of the feedstock materials used for composting and ensiling in both trials are given in Appendix Table A3. The leek residues from trial 1 had a lower OM ($p < 0.001$) and N ($p < 0.001$) content in comparison to the residues of white cabbage and the leek residues used in trial 2 and a lower P ($p = 0.002$) content compared to the residues of white cabbage. The higher OM content for leek in trial 2 (54% vs. 17% of DM in trial 1) could be explained by a lower soil particle load, i.e., less soil attached to the leek (possibly due to washing the leek before the outer leaves were removed while trimming the leek for transport to the market). The bulking agents (poplar wood chips, poplar bark and wheat straw used for composting in trial 1, and the chopped heath, strawberry substrate, rye straw and poplar bark used in trial 2), were richer in C than the crop residues, with a C/N ratio between 33.3 and 118.3 versus between 13.8 and 17.4 ($p < 0.001$), respectively. The bulking agents were also considerably drier than the fresh crop residues ($p < 0.001$). Furthermore, the maize straw used for composting in trial 1 was characterized by a low OM content, related to a high soil particle load. The maize straw used for ensiling was characterized by a high OM content, especially in trial 2 (91.6% of DM).

All composts had the lowest OM content (expressed on DM), followed by the silages and digestates (Table 3.2). The OM content of $LC2_{heath}$ and $LC2_{strawberry}$ was considerably higher

than LC1 and CC1. DM content was lowest for the digestates, followed by the silages and composts. Total N content was higher for LS1 and LS2 compared to LC1, $LC2_{heath}$, $LC2_{strawberry}$ and LD1. In trial 1, the C/N ratio of LS1 and CS1 was higher than the C/N ratio of the fresh vegetable residues due to the addition of the maize residues (Appendix Table A3). This was not the case in trial 2, because proportionally more leek was added and the maize straw had a lower C/N ratio than in trial 1. The C/N ratio of LD1 and CD1 (Table 3.2) was lower than the C/N ratio of the fresh leek residues (Appendix Table A3) and LC1, CC1, LS1 and CS1 (Table 3.2). In general, the silages had a lower pH-H_2O and a higher EC than the composts. Both composts from trial 1 were characterized by low mineral N concentrations. In trial 1, the NO_3^--N/NH_4^+-N ratios > 1 and OURs between 10-15 mmol kg^{-1} OM h^{-1} ('moderately stable' according to Flemish compost standards) indicated stable composts. However, the biodegradation potential did not decrease compared to the feedstock mixtures at the start (Appendix Table A3). The composts from trial 2 were more stable as reflected in the low OUR value (< 5 mmol kg^{-1} OM h^{-1} i.e. 'very stable' according to Flemish compost standards) and had biodegradation potentials that decreased from 2.5-3.0 to 1.0-1.4. In both trials, the silages were characterized by a lower relative lignin content (between 5.0 and 6.1% of OM) and a higher NH_4^+-N content (338-1179 mg kg^{-1} DM), OUR (78.6-136.8 mmol kg^{-1} OM h^{-1}) and biodegradation potential (11.3-17.2) than the composts (lignin: 23.5-40.3% of OM, NH_4^+-N: 0-45 mg kg^{-1} DM, OUR: 2.7-12.0 mmol kg^{-1} OM h^{-1} and biodegradation potential: 1.0-3.2). This indicates a low biochemical stability and expected high level of continued degradation of the silages when added to the soil, and a high biochemical stability of the composts and digestates of the ensilaged crop residues (between 1.7 and 1.9). $LC2_{strawberry}$ contained high concentrations (> 0.18 mg kg^{-1} compost) of certain fungicides (fludioxonil, boscalid, dimethomorph and tetraconazole) used during the cultivation of strawberry (Appendix Table A6).

Table 3.2 Characterization of composts, silages and digestates from trial 1 and 2 (mean ± standard deviation, n = 4). Significant differences between cabbage treatments in trial 1 are indicated with different italic letters, between leek treatments in trial 1 with different capital letters and between treatments in trial 2 with different bold letters (Scheffé test, $p < 0.05$; t-test for fresh bulk density and OUR between composts in trial 2). DM = dry matter, OM = organic matter, nda = no data available.

	Trial 1												Trial 2							
	Cabbage compost		Leek compost		Cabbage silage		Leek silage		Cabbage digestate		Leek digestate		Leek compost$_{heath}$		Leek compost$_{strawberry}$		Leek silage		Leek digestate	
pH-H_2O (-)	8.7 ± 0.1	*B*	8.6 ± 0.0	B	5.2 ± 0.1	*A*	4.2 ± 0.2	A	8.7		8.6		7.4 ± 0.0	**B**	7.8 ± 0.0	**C**	4.4 ± 0.1	**A**	nda	
Electrical conductivity (µS cm^{-1})	715 ± 34	*A*	548 ± 19	A	1728 ± 155	*B*	1047 ± 61	B	nda		nda		247 ± 11	**A**	340 ± 20	**A**	1090 ± 111	**B**	nda	
Fresh bulk density (kg m^{-3})	617 ± 5		563 ± 8		492		395		nda		nda		514 ± 3	**B**	492 ± 9	**A**	446		nda	
Organic matter content (% of DM)	27.4 ± 1.6	*A*	35.7 ± 2.0	A	32.7 ± 4.0	*B*	73.2 ± 5.8	B	71.9 ± 1.1	*C*	80.7 ± 1.7	B	47.9 ± 2.4	**A**	71.4 ± 1.2	**B**	89.1 ± 0.6	**C**	57.5	
Organic matter content (% of fresh weight)	12.7 ± 0.6	*C*	14.3 ± 0.7	C	8.3 ± 1.1	*B*	12.1 ± 1.3	B	4.5 ± 0.1	*A*	4.5 ± 0.1	A	15.0 ± 0.7	**B**	16.3 ± 0.4	**C**	9.5 ± 0.3	**A**	3.1	
Dry matter content (% of fresh weight)	46.5 ± 0.5	*C*	40.1 ± 0.5	C	25.6 ± 1.6	*B*	16.5 ± 0.6	B	6.3 ± 0.2	*A*	5.6 ± 0.1	A	31.4 ± 0.2	**C**	22.8 ± 0.5	**B**	10.6 ± 0.2	**A**	5.4	
NO_3^--N (mg kg^{-1} DM)	9 ± 2		4 ± 2		< 40		< 77		nda		nda		< 31		147 ± 7		< 106		nda	
NH_4^+-N (mg kg^{-1} DM)	0	*A*	0	A	1179 ± 328	*B*	802 ± 80	B	nda		nda		< 31		< 45		338 ± 50		nda	
NO_3^--N/NH_4^+-N (-)	> 1		> 1		< 0.0		< 0.1		nda		nda		-		< 3.3		< 0.3		nda	
Total N (g kg^{-1} DM)	7.07 ± 0.16	*AB*	7.93 ± 0.33	A	8.66 ± 1.42	*B*	13.39 ± 1.14	B	6.25 ± 0.07	*A*	7.02 ± 0.11	A	12.37 ± 1.42	**A**	14.08 ± 0.16	**A**	24.32 ± 0.94	**B**	nda	
Total P (g kg^{-1} DM)	1.69 ± 0.04	*A*	1.54 ± 0.20	A	1.64 ± 0.11	*A*	1.85 ± 0.17	A	nda		nda		1.14 ± 0.04	**A**	2.29 ± 0.08	**B**	3.17 ± 0.05	**C**	nda	
C/N (-)	21.5 ± 1.1	*B*	25.0 ± 2.1	B	21.1 ± 1.6	*B*	30.5 ± 3.4	C	11.2 ± 0.0	*A*	11.4 ± 0.1	A	21.6 ± 1.6	**A**	28.2 ± 0.3	**B**	20.4 ± 0.8	**A**	nda	
C/P (-)	91.0 ± 5.2	*A*	131.9 ± 20.6	A	110.4 ± 8.2	*B*	221.3 ± 28.8	B	nda		nda		232.4 ± 3.4	**C**	173.3 ± 4.4	**B**	156.4 ± 2.2	**A**	nda	
N/P (-)	4.2 ± 0	*A*	5.3 ± 1.0	A	5.2 ± 0.5	*B*	7.2 ± 0.1	B	nda		nda		10.8 ± 1.0	**C**	6.2 ± 0.2	**A**	7.7 ± 0.3	**B**	nda	
Oxygen uptake rate (mmol kg^{-1} OM h^{-1})	12.0 ± 2.4	*A*	10.0 ± 1.8	A	136.8	*B*	78.6	B	nda		nda		3.3 ± 0.1	**A**	2.7 ± 1.9	**A**	113.3		12.2	
Hemicellulose (% of OM)	29.8 ± 12.8	*A*	26.4 ± 4.8	A	40.4 ± 18.4	*A*	27.7 ± 1.8	A	13.4		15.7		17.4 ± 11.6	**A**	15.5 ± 6.1	**A**	22.3 ± 1.3	**B**	15.5	
Cellulose (% of OM)	51.4 ± 5.1	*A*	46.3 ± 4.2	A	63.8 ± 9.7	*A*	43.1 ± 1.3	A	10.9		10.9		24.1 ± 8.7	**A**	29.3 ± 4.7	**B**	34.6 ± 0.7	**C**	12.9	
Lignin (% of OM)	25.1 ± 0.8	*B*	23.5 ± 0.2	B	6.1 ± 0.9	*A*	5.5 ± 0.5	A	14.0		15.8		40.3 ± 4.7	**C**	32.9 ± 2.6	**B**	5.0 ± 0.2	**A**	14.9	
Biodegradation potential (-)	3.2 ± 0.6	*A*	3.1 ± 0.3	A	17.2 ± 4.3	*B*	13.0 ± 1.3	B	1.7		1.7		1.0 ± 0.4	**A**	1.4 ± 0.3	**A**	11.3 ± 0.5	**B**	1.9	
Total K (g kg^{-1} DM)	7.05 ± 0.31	*A*	6.24 ± 1.06	A	11.79 ± 1.52	*B*	16.84 ± 1.16	B	nda		nda		5.88 ± 0.36	**A**	10.66 ± 0.19	**B**	32.74 ± 1.12	**C**	nda	
Total Mg (g kg^{-1} DM)	1.67 ± 0.06	*A*	1.68 ± 0.19	A	1.65 ± 0.14	*A*	2.13 ± 0.16	B	nda		nda		1.43 ± 0.03	**A**	3.52 ± 0.09	**C**	2.75 ± 0.11	**B**	nda	
Total Ca (g kg^{-1} DM)	13.41 ± 0.65	*A*	12.85 ± 1.89	B	12.59 ± 1.96	*A*	4.90 ± 0.95	A	nda		nda		8.76 ± 0.53	**B**	22.80 ± 0.79	**C**	4.72 ± 0.18	**A**	nda	
Total Na (g kg^{-1} DM)	0.18 ± 0.02	*A*	0.20 ± 0.03	A	0.47 ± 0.25	*A*	0.51 ± 0.11	B	nda		nda		0.16 ± 0.01	**A**	0.38 ± 0.03	**C**	0.20 ± 0.01	**B**	nda	

3.3.2. Nitrogen mineralization experiments

3.3.2.1. pH-KCl

At the start of incubation, all treatments had a significantly higher soil pH than the control soils without amendments, except LC1, $LC2_{heath}$ and $LS1_{60N}$. In the anaerobically digested silages the pH rose strongly immediately after soil incorporation (up to 7.5 ± 0.01 for LD2). In the following days, soil pH decreased, with the digestates reaching lowest pH levels, then remained constant after approximately two weeks. At the end of the incubation period LC1 (5.60 ± 0.03), CC1 (5.63 ± 0.05), $LS1_{60N}$ (5.54 ± 0.09), $CS1_{60N}$ (5.65 ± 0.02) and $LC2_{strawberry}$ (6.09 ± 0.02) had a significantly higher soil pH compared to the control soil (5.36 ± 0.01, 5.18 ± 0.01 and 6.00 ± 0.03 for CON1, $CON1_{60N}$ and CON2, respectively), while LD2 (5.59 ± 0.02) and FL (5.73 ± 0.02) had a significantly lower soil pH compared to the control soil (6.00 ± 0.03).

3.3.2.2. N mineralization

A linear increase of mineral N in the control soil was observed during the incubation period (data not shown). Directly after digestate application (day 0), we found a significantly higher mineral N content compared to after compost application (Figure 3.1). The N mineralization rates were calculated using zero order kinetics, i.e., $N(t) = N_{min,0} + k*t$; where the intercept $N_{min,0}$ is the net mineral N content in the soil amended with composted, ensilaged or digested crop residue with or without N fertilizer at time t = 0 and k is the mineralization rate. For the composts and digestates in both trials, the slope of the linear regression line was not significantly different from zero, which means no significant N mineralization nor immobilization occurred, except for a small N immobilization in $LC2_{heath}$ observed on day 21, 56 and 91 (max. -1.2 mg N kg^{-1} soil). In contrast to the composts and digestates, N immobilization occurred in $LS1_{60N}$ and $CS1_{60N}$ (Figure 3.1A). N immobilization remained fairly constant until day 69 (on average -18.6 mg N kg^{-1} soil for $CS1_{60N}$ and -29.3 mg N kg^{-1} soil for $LS1_{60N}$ from day 15 until 69), after which $N_{min,net}$ tended to become positive again. Similarly, $LS2_{60N}$ showed net N immobilization in the beginning up to 36 days of incubation ranging from -6.5 to 0 mg N kg^{-1} soil, after which net N mineralization appeared. $LS2_{60N}$ showed a mineralization rate of 0.29 mg N kg^{-1} soil day^{-1} ($p < 0.001$, $R^2 = 0.70$) (Figure 3.1B). FL showed a rapid increase of N mineralization and continued to mineralize during the entire incubation period with a mineralization rate of 0.48 mg N kg^{-1} soil day^{-1} ($p = 0.016$, $R^2 = 0.72$) (Figure 3.1B).

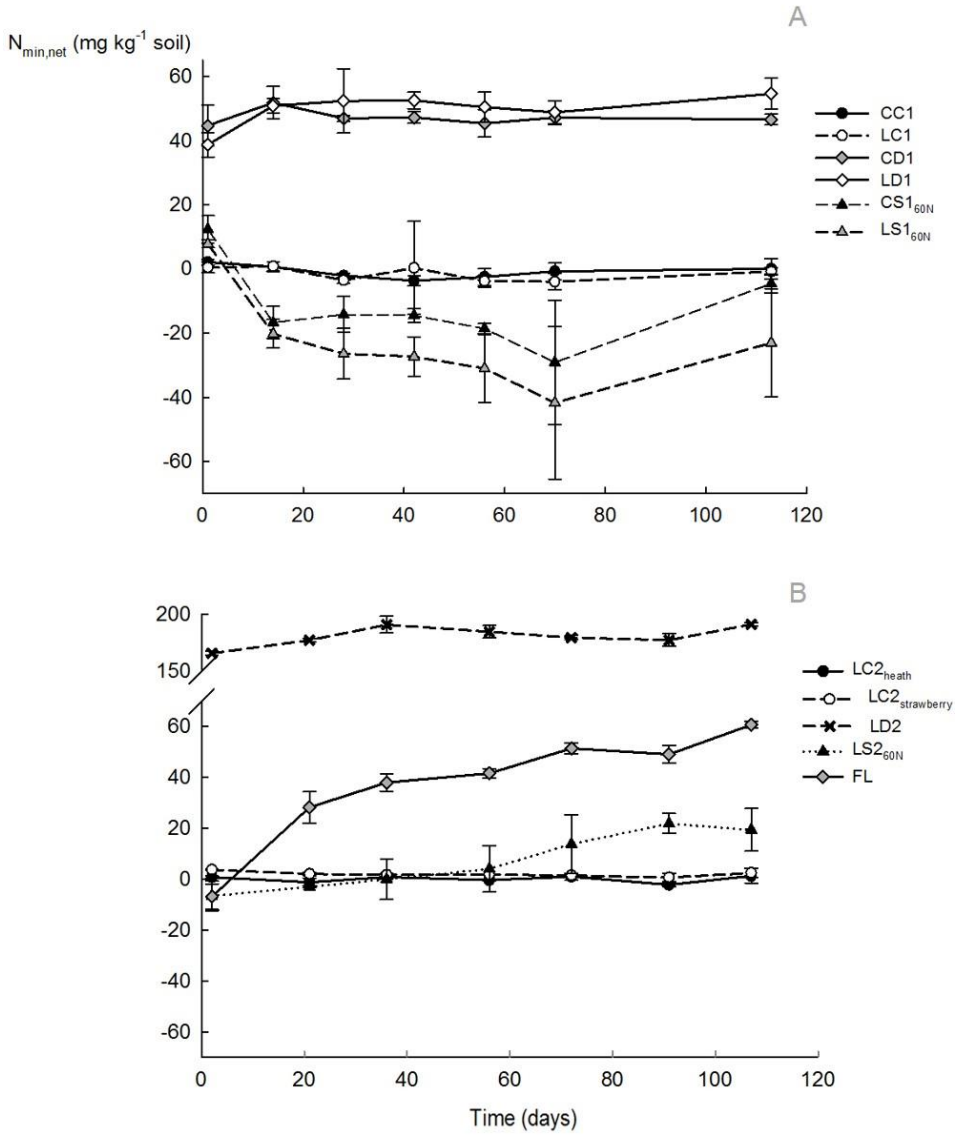

Figure 3.1 Net N mineralization ($N_{min,net}$) in end products from trial 1 (A), with CC1 = cabbage compost, LC1 = leek compost, CD1 = cabbage digestate, LD1 = leek digestate, $CS1_{60N}$ = cabbage silage + 60N, $LS1_{60N}$ = leek silage + 60N; and trial 2 (B), with $LC2_{heath}$ = leek compost with chopped heath biomass, $LC2_{strawberry}$ = leek compost with strawberry substrate, LD2 = leek digestate, $LS2_{60N}$ = leek silage + 60N, FL = fresh leek. Error bars indicate standard deviations (*n* = 3).

3.3.2.3. Microbial biomass C

In both trials, the strongest increase in microbial biomass carbon (MBC) took place when ensilaged residues with additional N fertilization were applied. The increase in MBC at 6-8 weeks after application of the processed residues was about twice as high for the ensilaged residues compared to the digested and composted residues (Table 3.3). Incorporation of FL also resulted in a significantly higher MBC compared to the addition of the composted or ensilaged residues.

Table 3.3 Microbial biomass carbon (mean ± standard deviation, n = 3) after 6 and 12 weeks (trial 1) or 8 and 13 weeks (trial 2) of incubation for the treatments of the N mineralization experiment. Groups of treatments with the same letter are not significantly different per sampling time (Scheffé test, $p < 0.05$). With CON = control soil without amendment, CON_{60N} = control soil + 60N, CC1 = cabbage compost, LC1 = leek compost, $CS1_{60N}$ = cabbage silage + 60N, $LS1_{60N}$ = leek silage + 60N, CD1 = cabbage digestate, LD1 = leek digestate; $LC2_{heath}$ = leek compost with chopped heath biomass, $LC2_{strawberry}$ = leek compost with strawberry substrate, $LS2_{60N}$ = leek silage + 60N, LD2 = leek digestate, FL = fresh leek.

	Microbial biomass carbon (µg C g^{-1} dry soil)								
	TRIAL 1					TRIAL 2				
	6 weeks		12 weeks			8 weeks		13 weeks	
CON1	28.1 ± 4.6	a	32.5 ± 2.5	a	CON2	39.3 ± 1.6	a	33.0 ± 8.3	a
$CON1_{60N}$	29.3 ± 2.6	ab	33.4 ± 7.0	a	$CON2_{60N}$	37.2 ± 1.6	a	32.3 ± 4.2	a
CC1	42.4 ± 2.8	c	56.0 ± 9.2	bc	$LC2_{heath}$	46.7 ± 3.2	a	52.4 ± 3.2	a
LC1	40.2 ± 3.3	bc	43.1 ± 1.9	ab	$LC2_{strawberry}$	51.2 ± 4.9	a	43.8 ± 8.6	a
$CS1_{60N}$	78.9 ± 2.5	d	72.1 ± 3.0	cd	$LS2_{60N}$	92.9 ± 9.4	b	75.2 ± 12.5	b
$LS1_{60N}$	81.8 ± 5.3	d	77.6 ± 5.2	d	LD2	43.9 ± 9.5	a	43.9 ± 4.6	a
CD1	32.7 ± 2.3	abc	33.5 ± 2.6	a	FL	73.4 ± 3.3	b	61.8 ± 6.1	b
LD1	39.9 ± 2.8	abc	36.4 ± 1.7	a					

3.3.3. Carbon mineralization experiments

Mineral N did not limit C mineralization from the silage, since enough soil mineral N was available and adding extra N did not increase C mineralization compared to silage without extra N (data not shown). Generally, throughout the incubation period the relative C mineralization ($C_{min,net}$) was the highest for FL (up to 90% of the added C after 82 days), followed by the ensilaged (between 30 and 47%), digested (19-27%) and composted (2-21%) crop residues, respectively (Figure 3.2). In trial 2 (B), both composts showed a lower $C_{min,net}$ compared to trial 1. For FL, silages and digestates, a rapid increase in C mineralization rate was observed from the start up to two weeks of incubation, but in succeeding weeks the mineralization rate decreased. The composts showed a more linear trend in C mineralization with a mineralization rate between 0.76 and 0.84 µg g^{-1} soil. The easily decomposable C pool (C_{af}) was high in FL (776 ± 29 µg g^{-1} soil) and in the silages (between 581 and 992 µg g^{-1} soil), with lower values in the digestates (between 111 and 427 µg g^{-1} soil).

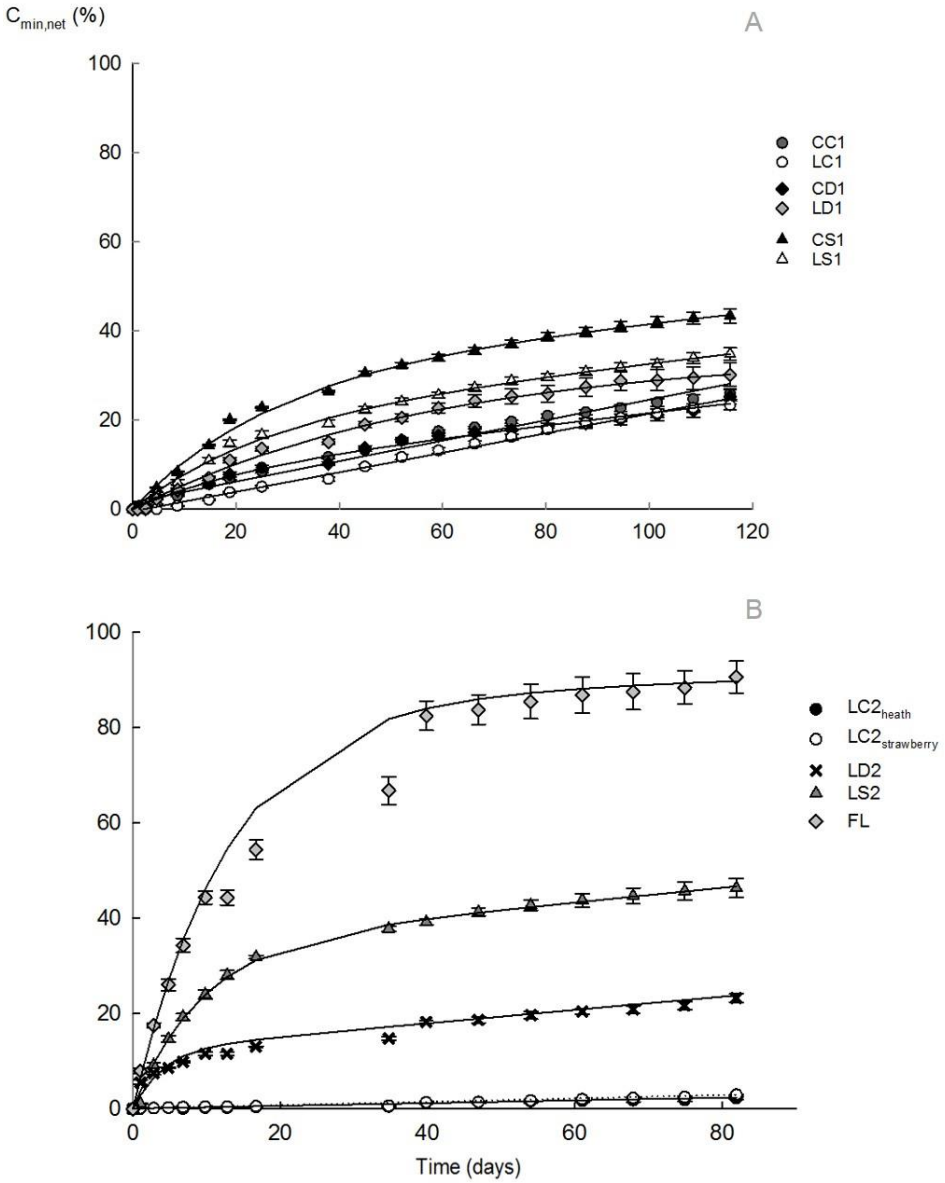

Figure 3.2 Cumulative amount of net mineralized C ($C_{min,net}$ in % relative to the C input) across the incubation period from trial 1 (A), with CC1 = cabbage compost, LC1 = leek compost, CD1 = cabbage digestate, LD1 = leek digestate, CS1 = cabbage silage, LS1 = leek silage; and trial 2 (B), with $LC2_{heath}$ = leek compost with chopped heath biomass, $LC2_{strawberry}$ = leek compost with strawberry substrate, LD2 = leek digestate, LS2 = leek silage, FL = fresh leek. The symbols represent measured values, the lines modelled values. Error bars indicate standard deviations (n = 3).

3.3.4. N_2O emission experiment

Figure 3.3 shows the cumulative net N_2O emissions after amending FL and the end products from trial 2. The highest N_2O emission rates were observed during the first 3 days (except for LD2, which remained high after 7 days) and were negligible from day 12 onwards, except for FL. The addition of 60 kg N ha^{-1} ($LS2_{60N}$, data not shown) increased N_2O emissions by more than four times (69.7 for LS2 and 304.7 ng N_2O-N g^{-1} soil for $LS2_{60N}$). Cumulative

N_2O emissions from LS2 and LD2 (76.2 ng g^{-1} soil) were higher than from FL (16.3 ng g^{-1} soil). Cumulative N_2O emissions of the compost amendments were found to be the lowest (6.6 and 5.5 ng g^{-1} soil for $LC2_{heath}$ and $LC2_{strawberry}$, respectively).

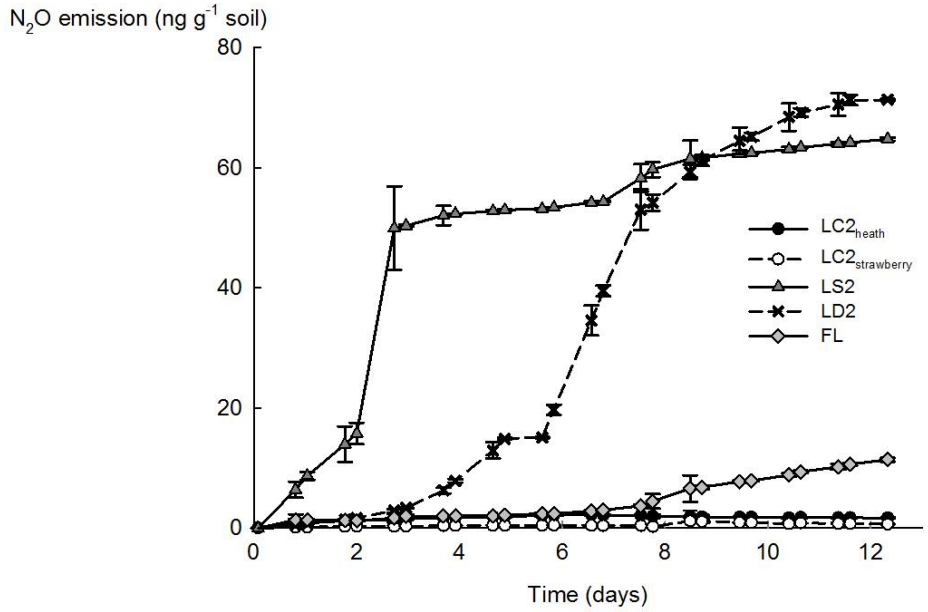

Figure 3.3 Cumulative net N_2O emissions (ng g^{-1} soil) in end products from trial 2, with $LC2_{heath}$ = leek compost with chopped heath biomass, $LC2_{strawberry}$ = leek compost with strawberry substrate, LS2 = leek silage, LD2 = leek digestate, FL = fresh leek. Error bars indicate standard deviations (*n* = 3).

3.4. Discussion

3.4.1. Product quality and effect on soil pH

Processing N-rich cabbage and leek residues by co-composting, AD and co-ensiling of those silages resulted in different end products related to the different degradation processes. In case of composting the crop residues, C-rich materials were mixed with N-rich crop residues, leading to an aerobic biodegradation process and increased temperatures, hence resulting in a drier and more stable end product with less, but more recalcitrant OM and less mineral N compared to silages and digestates. Note the importance of minimizing the soil particle load of the crop residues before composting, since the composts from trial 1 (where soil particle load of the residues was high) had a considerably lower OM content and stability compared to the composts from trial 2, as indicated by a higher OUR, no decrease in biodegradation potential during the process and a remaining high biodegradation potential (3.1-3.2) in the composts from trial 1. In other studies, the biodegradation potential decreased from 1.7-7.6

to 0.7-2.5 during composting with different feedstock, i.e., different types of manure, green waste and household waste (Blanco and Almendros, 1997; Eklind and Kirchmann, 2000; Francou et al., 2008; Lashermes et al., 2012; Vandecasteele et al., 2014). Francou et al. (2008) also reported a high biodegradation potential (2.5) for a compost (initial dry mass consisting of 59% paper-cardboard) explained by a slower composting process due to N shortage. This could also explain the high biodegradation potential of the composts from trial 1, since they had low mineral N contents, indicative of a N shortage during composting. Ensiling, an anaerobic conservation process with low OM degradation, results in an unstable end product rich in easily decomposable C and NH_4^+-N, in accordance with our first hypothesis. Further processing of the silage by AD resulted in decomposition of the easily mineralizable C compounds, leading to a stable digestate that maintained a high mineral N content.

During ensiling, water-soluble carbohydrates are fermented by anaerobic bacteria into organic acids, as a result, the silages were characterized by high NH_4^+-N concentrations and a slightly acidic pH (< 5.2) in comparison to the composts and digestates. In spite of this, after soil amendment with the silages, the soil pH-KCl did not decrease and even increased compared to the control soil after 113 days. This observation confirms the findings of Agneessens et al. (2015), who stated that the soil pH buffering capacity and buffering agents in the silages can explain results such as these. The opposite effect was observed when amending the soil with fresh leek residues and especially digestate, both of which caused the soil pH to drop after 113 days. These results can be explained by the nitrification process during degradation in the soil. Except for $LC2_{heath}$, incorporation of compost at the dose applied in this study increased the soil pH after 113 days, as observed in other studies (D'Hose et al., 2016).

3.4.2. Effect of product amendment on soil N and C dynamics

To the best of our knowledge, this paper is the first to compare how different methods of processing the same vegetable crop residues result in products with different levels of stability and greatly different effects on soil N and C dynamics. The highest MBC and rate of net C and N mineralization was observed when applying non-processed FL, as the residues started to degrade in the soil directly after application. During co-ensiling and co-composting, the fresh crop residues were mixed with materials rich in C, which resulted in lower net N and C mineralization rates compared to the incorporation of FL, even though the products were added to a base with equal amounts of N. The lower N mineralization rates

when incorporating processed crop residues result from an increase in C/N ratio by mixing the crop residues with biomass with a higher C/N ratio. Generally, the highest net C mineralization was observed in FL, followed by silages, then digestates, with composts having the lowest value. Furthermore, the C mineralization rate of the composts was linear according to their stable character, while the FL, silages and digestates showed a rapid increase in the beginning, followed by a decrease.

Ensiling crop residues does not further stabilize them, thus they continue to degrade after being incorporated into the soil. This was reflected by a strong increase in MBC and C mineralization, a rapid N immobilization when incorporating the silages from trial 1 ($LS1_{60N}$ and $CS1_{60N}$) and higher N_2O emissions compared to compost application. This confirmed our second hypothesis and agrees with a previous study (Agneessens et al., 2015) where reapplication of silages to soil promoted microbial activity. In that study 40-48% of the C added with the silages was mineralized after 100 days, leading to a temporary N immobilization. N immobilization was lower when $LS2_{60N}$ was applied, as explained by its lower C/N ratio compared to $LS1_{60N}$ (20.4 vs. 30.5). Moreover, for $LS2_{60N}$, remineralization occurred after 36 days, induced by the enhanced microbial activity (larger increase in MBC and C mineralization rate) (Agneessens et al., 2015). N_2O emissions were higher after silage amendment compared to after amending composts, digestate or fresh leek in the first 7 days, possibly explained by the higher C and N availability in the silage (Huang et al., 2004b). However, compared to the study of Huang et al. (2004b), the N_2O emissions were low. In their study, they found cumulative emissions between 384 and 568 ng g^{-1} soil after 21 days, after amending sugarcane stalk and rapeseed cake, respectively (under the same incubation circumstances as in our study). Moreover, when expressing the N_2O emissions in kg ha^{-1}, only 0.10 kg N_2O-N ha^{-1} was lost after 12 days, which is lower than reported emissions for lettuce residues between 0.34 - 1.42 kg N_2O-N ha^{-1} (values dependent on the tillage practice; experiments performed on a freely drained loamy sand soil under field conditions) after 79 days (majority of the emissions were in the first 14 days) (Baggs et al., 2000).

Incorporation of composts did not lead to net N mineralization and showed the lowest N_2O emissions (0.001-0.003 kg ha^{-1}), as the bulk (> 90%) of the C and total N in compost is present in a (stabilized) organic form, which decreases the risk for N leaching to soil and surface waters (Amlinger et al., 2003), and part of the N could be already lost during the composting process. Total volatile N losses during composting (mainly ammonia, and to a lesser extent N_2O) are strongly dependent on the feedstock composition and composting

circumstances, and can amount to 42% of the initial N content when composting cattle manure under suboptimal conditions (Eghball et al., 1997). However, in our experiment, indicative mass balances showed that N losses during composting were below 20% of the initial N content. Other studies reported N immobilization after compost amendment, related to the decomposability of C and N of the residues (Galvez et al., 2011). Furthermore, composts show low C mineralization (2.4-20.5% of the added C) after soil application, as previously reported (e.g. between 1.8 and 8.8% of the total added organic C on a DM basis in the study of De Neve et al. (2003) and < 5% in the study of Galvez et al. (2011)), and thus help to increase the OM content of the topsoil (D'Hose et al., 2016). This result was in agreement with our third hypothesis. Incorporation of $LC2_{strawberry}$ and $LC2_{heath}$ (the composts from trial 2) led to a much lower cumulative C mineralization after 82 days (2.4 - 3.0%) compared to the composts from trial 1 (17 - 20%). This could be related to their higher stability and more microbial resistant OM. In the study of Garcia-Gomez et al. (2003), 5.4% of TOC mineralized after 70 days in mature compost, in contrast to 25% for immature compost. Moreover, even after the composting process, certain fungicides (fludioxonil, boscalid, dimethomorph and tetraconazole) used during the cultivation of strawberry were still detectable in the compost and could inhibit fungi responsible for further degradation. The low C mineralization in $LC2_{heath}$ could be explained by large quantities of soluble phenolic compounds in the chopped heath biomass, enhancing recalcitrant humic complexes through condensation and microbially mediated immobilization of organic N during degradation (Kristensen and McCarty, 1999; Kuiters, 1990).

Soil amendment of digestates resulted in the highest soil mineral N content and cumulative N_2O emissions and did not lead to N immobilization, as observed by Fuchs et al. (2008). A larger amount of available mineral N in the digestate during incubation might have contributed to higher N_2O emissions compared to composts and fresh leek. The lower emissions compared to the silage in the beginning of the incubation period could be explained by the lower C availability in the digestate (Huang et al., 2004b). Again, when expressing the N_2O emissions in kg ha^{-1}, only 0.11 kg N_2O-N ha^{-1} was lost after 12 days, which is generally low and might be related to the low NO_3^--N content or C availability (Bremner, 1997). It is not yet known how soil N_2O emissions are affected when applying digestate based on silages of crop residues. Moreover, the emissions are strongly dependent on the availability of NO_3^--N, available organic C, O_2 and soil texture, and temperature (Bremner, 1997). Our primarily findings therefore call for a more detailed investigation.

Furthermore, digestate application led to C mineralization between 19 and 27% of the added C, comparable to the study of de la Fuente et al. (2013). As we hypothesized, C mineralization of digestate is lower than of fresh leek and silages, according to the more stable character of the digestates: because the most labile C fractions are converted into methane and CO_2 during the digestion process, the less readily available C remains in the digestate (Thomsen et al., 2013). However, Thomsen et al. (2013) calculated that the long-term retention of plant derived C in soil is little affected by pretreatments such as passage through a ruminant and AD.

3.4.3. OUR and biodegradation potential as indicators for C mineralization

C mineralization experiments are time-consuming, thus we tried to correlate the cumulative C mineralization of the products after 80 days with other, less labor-intensive or faster stability parameters such as C/N ratio, OUR and biodegradation potential. We found no significant relation between C mineralization and C/N ratio, but we did find a log-linear relation with biodegradation potential (C_{min} = 12.80 * ln(biodegradation potential) + 5.02; R^2 = 0.82; $p < 0.01$) and OUR (C_{min} = 9.37 * ln(OUR); R^2 = 0.91; $p < 0.01$). This implies that OUR and biodegradation potential were good parameters for estimating cumulative C mineralization in our study. OUR and biodegradation potential can thus be used for quickly screening the products, but more data is necessary for a robust prediction of the C mineralization rate. The good correlation between C mineralization rate and biochemical composition had been reported previously (Francou et al., 2008); that study also contains a dataset of the initial feedstock materials and feedstock mixtures, and the composts at the end of the process. However, the measurement of OUR can be very sensitive to changes of moisture, temperature, oxygen and N availability (Veeken et al., 2007). For example, N shortage will slow down the degradation process, resulting in an apparent low OUR, indicating a stable product when this is not the case. Hence, we want to highlight that it is necessary to evaluate a set of stability indicators to have a reliable idea about the stability of the product.

Figure 3.4 shows that the products were clearly distinctive in stability, with silages being the less stabilized products (OUR > 78.6 mmol kg^{-1} OM h^{-1}, biodegradation potential > 11.3 and C mineralization > 30%), followed by the more stable digestates (OUR < 12.2 mmol kg^{-1} OM h^{-1}, biodegradation potential < 1.9 and C mineralization < 27%) and the most stable

composts from trial 2 (OUR < 3.3 mmol kg^{-1} OM h^{-1}, biodegradation potential < 1.4 and C mineralization < 3%). The stability of the composts from trial 1 was lower than from trial 2 and were comparable to the digestates, since the composting process was less optimal (see section 3.4.1 above).

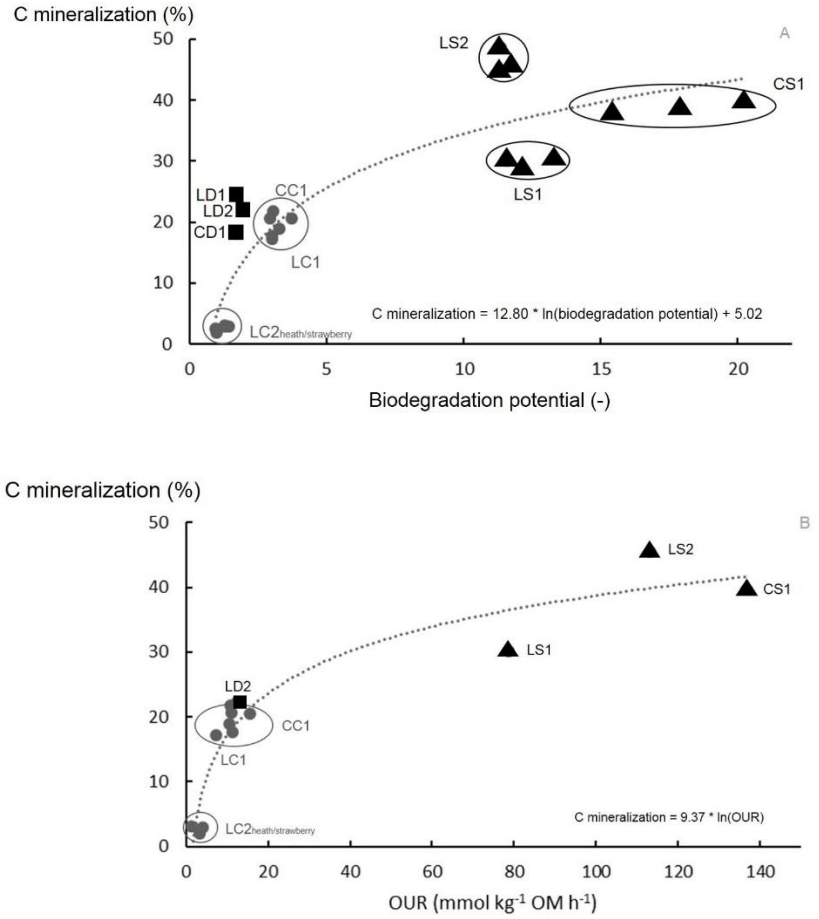

Figure 3.4 Log-linear relationship between the cumulative amount of net mineralized C (in % of added C after 82 days) and (A) biogedradation potential, (B) oxygen uptake rate (OUR) for silages (triangles), composts (spheres) and digestates (squares).

3.4.4. Practical implications

Co-ensiling, co-composting and anaerobic co-digestion of N-rich crop residues, already available on-farm (e.g., leek residues after washing and cleaning, or after field removal) are appropriate techniques to avoid N losses. These techniques also close nutrient and material cycles on-farm when the end products are reapplied on the field. Depending on the context-specific soil and crop requirements the different characteristics of each of these products can allow for a more flexible application in time and space than use of fresh crop residues.

Ensiling is a low-cost conservation technique enabling a large flexibility concerning storage and application of nutrients and OM (Agneessens et al., 2015). However, direct application of ensilaged crop residues causes temporary N immobilization, especially for silages with a high C/N ratio and even when extra fertilizer is added. Therefore, to avoid N deprivation for the following crop, a large enough time interval between field incorporation of the silages and crop sowing date should be considered (Agneessens et al., 2015) (e.g., by already applying them in autumn as soil improving amendment), and/or the silages could be combined with fast-release fertilizers, but field trials are needed to assess this.

While ensiling is a conservation technique with limited degradation and N assimilation after soil incorporation, composting is a controlled degradation and stabilization process where N is mainly assimilated during the composting process, resulting in a product with stable C pools. In this way, the produced compost can be applied as a valuable soil improver and slow-release organic fertilizer. It has been demonstrated previously that composting might have the additional advantage of resulting in the destruction of weed seeds (Eghball and Lesoing, 2000) and/or pathogens (Lung et al., 2001) when sufficiently high temperatures are reached during the composting process. Moreover, microbial antagonism and/or competition for nutrients, toxicity from byproducts of organic matter decomposition (e.g., ammonia, sulfides, organic acids, and phenolic compounds) and enzymatic breakdown during the composting process are other aspects contributing to sanitation of the feedstock compared to the application of fresh residues (Wichuk et al., 2011). Whether the pH decrease during ensiling could result in a similar destruction, has not yet been studied. Studies on maize report that ensiling can limit the majority of fungal growth by the low oxygen content and pH, but some species can survive the silage environment (Mansfield and Kuldau, 2007). More research is necessary to test the survival of pathogens and weed seeds during ensiling of crop residues. Expertise about the composting practice is required, as the composting process should be carefully monitored and properly executed to produce stable compost that will not further degrade, but will rather contribute to soil C sequestration. Compared to ensiling, composting is a more labor intensive treatment and demands an extra financial investment, because it requires specific tools to monitor and manage the compost piles (Viaene et al., 2016b).

Furthermore, to ensure a good composting process, bulking agents with a higher C/N ratio should be added (Huang et al., 2004a), while a proper ensiling process requires mixing drier biomass with fresh crop residues (Yahaya et al., 2002). This implies that the biomass (i.e.

woody material for composting and straw for ensiling) must be readily available for the farmer, which is not always the case (Viaene et al., 2016b). We investigated the feasibility of ensiling pure leek residues without maize straw, and found double the amount of sap and N losses (sap loss of 25% of the initial fresh weight, and N loss with the sap of 11.6% of the initial N content of the silage) compared to co-ensiling with 40% v/v maize straw (sap loss of 13% of the initial fresh weight, and N loss with the sap of 6.5%). Another point of attention for composting and ensiling is to minimize the soil particle load of the crop residues to ensure a high OM content of the end product.

After ensiling, the silages can be used as AD feedstock to produce bioenergy, after which the remaining stabilized digestate can be reapplied to the soil. Compared to compost, digestate contains more plant-available nutrients (mineral N) and therefore acts as a short-term fertilizer. The short-term N_2O losses after soil application of silages and digestates were higher compared to compost application, but were still generally low (< 0.11 kg ha^{-1}). AD implies either investing in an on-farm digester or transport from the farm to the central digestion plant and back.

3.5. Conclusions

This study clearly indicates the potential of co-ensiling, co-composting and application of digestate based on vegetable crop residues as promising alternatives for soil incorporation of fresh vegetable crop residues after harvest. Those valorization options can be used to process and/or store the crop residues during winter, thus reducing the risk for N losses due to the fast decomposition of fresh crop residues. Furthermore, by reapplying silages, composts and digestates to the soil, the soil quality benefits and nutrient cycles are closed on a local scale. Compost, digestate and silage have different characteristics and, when applied to soil, lead to product-specific C and N dynamics. Soil amendment of silages that are still highly biodegradable resulted in highest C mineralization and MBC and temporary N immobilization, whereas application of mature composts led to low C mineralization and no net N mineralization or immobilization. The effect of digestate application was comparable to compost application, while higher soil mineral N and N_2O emissions were found. The on-farm facilities and different soil effects of processed vegetable crop residues should be taken into account when choosing the most appropriate valorization option and application in time and space.

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CHAPTER 4

Potential of alternative bulking agents for composting leek residues

After: Viaene, J., Reubens, B., Willekens, K., Van Waes, C., De Neve, S., Vandecasteele, B., 2016. Potential of chopped heath biomass and spent growth media to replace wood chips as bulking agent for composting vegetable crop residues. Manuscript submitted for publication.

Abstract

We investigated the potential of C-rich byproducts to replace wood chips as bulking agents (BA) during composting. The impact of these alternatives on the composting process and on compost stability and characteristics was assessed. Three BA (chopped heath biomass and spent growth media used in strawberry and tomato cultivation) were used for processing leek residues in windrow composting. All BA resulted in stable composts with an OM content suitable for use as soil amendment. Using chopped heath biomass led to high pile temperatures and OM degradation and a nutrient-poor compost with high C/P ratio appropriate for increasing soil organic carbon content in P-rich soils. Spent substrates can replace wood chips, however, due to their dense structure and lower biodegradation potential, adding a more coarse BA is required. Generally, the nutrient content of the composts with growth media was higher than the composts with wood chips and chopped heath biomass.

4.1. Introduction

Composting is a biological process in which microorganisms convert fresh organic material into a stable and humus-rich product under controlled conditions, i.e., optimal conditions of moisture and oxygen (Bokhorst and ter Berg, 2001). In order to ensure a good composting process, feedstock materials characterized by a high moisture content and a low C/N ratio should be co-composted with drier, C-rich bulking agents (BA) to provide structure and porosity to the mixture. These characteristics ensure proper aeration (Barrington et al., 2002), decrease the moisture content and supply biodegradable C. Sufficient C levels lead to microbial immobilization of N during the composting process, thereby minimizing N losses (Nolan et al., 2011) as well as generating enough metabolic heat to reach thermophilic temperatures and sanitize the compost. Commonly-used BA such as wood chips or tree bark are well-known and well-studied (e.g. Larney et al., 2008; Suzuki et al., 2004; Vandecasteele et al., 2013), but these wood-based BA may become more scarce and more expensive, causing a barrier to on-farm composting (Viaene et al., 2016b) and they can contain high concentrations of heavy metals (Vandecasteele et al., 2013).

Some studies, mainly from Mediterranean regions, have tested other byproducts for use as BA, including sugar cane bagasse (Meunchang et al., 2005), rice hulls (Leconte et al., 2009) and brans (Chang and Chen, 2010), grape stalks (Cayuela et al., 2010), olive leaves (Alburquerque et al., 2006), sawdust (Huang et al., 2004a), biochar (Dias et al., 2010), paper-cardboard (Francou et al., 2008) and straw (Michel et al., 2004) for co-composting kitchen waste and various types of manure. However, byproducts from nature management and greenhouse cultivation have been rarely tested as alternative BA. A potential byproduct from nature management is chopped heath biomass, obtained by a less intensive sod-cutting procedure for removing excess nutrients in heathland management. This process is gaining more interest as it has several advantages over standard sod-cutting (Niemeyer et al., 2007). The chopped biomass consists of above-ground vegetation and a large part of the O-horizon(s). Approximately 87 t fresh material ha^{-1} can be annually harvested (Viaene et al., 2014). In greenhouse cultivation, substrate reutilization is strongly encouraged because the disposal of growth media at the end of the growing season is a potential threat to the environment (Diara et al., 2012). Spent substrates (annually ca. 35 t ha^{-1}, Viaene et al. (2014)) can either be recycled as a soil amendment or mixed with other substrates after composting (Diara et al., 2012).

The main objective of this study is to test C-rich byproducts for their value as BA with the aim of replacing wood chips for co-composting N-rich vegetable crop residues. N-rich vegetable crop residues, such as leek residues generated during the cleaning process of leek, pose a high risk for N losses when applied to or left on the field during autumn (Chaves et al., 2007). Management of those crop residues is an effective strategy to reduce N losses, notably nitrate leaching and probably N_2O emissions (de Ruijter et al., 2010). On-farm composting of vegetable residues is a valuable management option. Not only would it reduce N losses, it promotes the local recycling of nutrients and OM and the production of a soil-improving compost (Viaene et al., 2016b). We tested three locally available alternative BA, namely chopped heath biomass and spent strawberry and tomato substrates. The performance of each BA was evaluated for its ability to ensure an optimal composting process (OM degradation, sufficiently high N immobilization, particle size distribution, sufficiently high pile temperatures for hygienic reasons) resulting in a mature compost (microbial and biochemical stability, low N immobilization) with soil improving characteristics (pH, EC, bulk density, OM and nutrient content, N fertilizer replacement value). To the best of our knowledge, this is the first study on composting of chopped heath biomass or spent substrates from greenhouse cultivation and the evolution of N immobilization capacity during the composting process.

4.2. Materials and Methods

4.2.1. Composting process

The compost trial was executed in open air on a concrete pad at the composting facility of the experimental farm of the Institute for Agricultural and Fisheries Research (ILVO) in Merelbeke, Belgium. On January 16th 2014 four windrows were set up; in the beginning of the process, each windrow was 12 m long, 3 m wide and 1.5 m high. Each windrow consisted of 200 kg of fresh straw and 4225 kg fresh weight (FW) of leek crop residues, collected at the farm after harvested leek was cleaned and prepared for the fresh market. Furthermore, each windrow contained an equal volume (20.8 m³) of the BA: wood chips from poplar, chopped heath, strawberry or tomato substrate. The strawberry substrate consisted of 50% peat, 35% coconut fibre and 15% perlite (Research Centre Hoogstraten, Hoogstraten, Belgium), the tomato substrate consisted of 40% peat sods, 20% coconut fibre and 40% peat mulch (Peltracom, Gent, Belgium). The materials were mechanically mixed with a tractor-

pulled windrow compost turner (TG 301, Gujer Innotec AG, Mesikon, Switzerland). The compost piles (further referred to as *Cw*, *Ch*, *Cs* and *Ct* for the pile with wood chips, chopped heath biomass, spent strawberry substrate and spent tomato substrate, respectively) were covered with a geotextile (TopTex® TenCate, Almelo, The Netherlands) to protect them from precipitation, while still allowing oxygen exchange. During the composting process, temperature (GHM Messtechnik GmbH Standort Greisinger, Regenstauf, Germany) and CO_2 levels (CO_2-Indicator TESTORYT, BRIGON Messtechnik GmbH, Rodgau, Germany) were monitored at least twice per week as the average of four point measurements along the length of the piles. When the critical temperature and/or CO_2 concentration (65°C and/or 16% CO_2) were reached, the piles were aerated by turning with a compost turner and water was added when the mixture became too dry (TG 301, Gujer Innotec AG, Mesikon, Switzerland). During the experiment, we took the temperature of the piles to determine whether the tested BA were successful in increasing the pile temperatures for a proper composting process. In all three windrows, the following additions were necessary: straw (200 kg FW/windrow) on day 5, a second dose of leek crop residues (2250 kg FW/windrow) on day 19, and additional poplar bark (2500 kg FW/windrow) on day 28. On day 34, the piles were compacted by moving the compost at the ends to the middle of the pile and turning the mixture. Before mixing the materials, individual feedstock materials were sampled. Samples of the feedstock mixtures and mature composts were taken at day 1, 14, 27, 55 and 76 of the composting process. At time of sampling, four mixed bulk samples, each consisting of 10 subsamples (± 3 L), per material or windrow were taken and sieved over a 10 mm mesh and stored at 4°C. Daily average temperature and precipitation data were collected from the weather station located at ILVO, where the composting facility is located.

4.2.2. Physico-chemical analyses

All samples were analyzed for physico-chemical quality parameters (< 10 mm fraction): fresh bulk density and dry matter (DM) content at 105°C (EN 13040), organic matter (OM) and ash content by mass loss during ashing at 550°C (EN 13039), pH-H_2O (EN 13037), electrical conductivity (EC) (EN 13038) and total N content (Dumas EN 13654-2, Thermo scientific – flash 4000 total N analyzer, Thermo scientific).

To evaluate the biodegradation process during composting, extra parameters were measured on day 14, 27, 55 and 76 (< 10 mm fraction). The particle size distribution was determined by sieving all samples through a 10 mm mesh and weighing the two fractions. As an indicator of stability, oxygen uptake rate (OUR) was calculated from the oxygen consumption due to

microbial activity. In a 1 L Schott flask, 20 g product and 200 mL buffered nutrient solution (with N-allylthiourea (6.25 mg per flask) as nitrification inhibitor and a phosphate buffer to buffer the suspension at pH 6.5) were shaken for five days at 120 rpm in a closed OxiTop respirometer at 20°C based on the method reported in Grigatti et al. (2011). The pressure course in the bottle headspace was continuously recorded by means of the OxiTop device (WTW GmbH, Germany), in which the CO_2 was trapped by Sodalime (Merck); the decrease in pressure under these conditions can only be due to the oxygen consumption from the microbial respiration. OUR was expressed as mmol kg^{-1} OM h^{-1}. Furthermore, neutral detergent fiber (NDF), acid detergent fiber (ADF) and acid detergent lignin (ADL) content in the dried feedstock and compost samples were determined with an Ankom220 Fiber Analyzer extraction unit according to the method of Van Soest et al. (1991). Based on NDF, ADF and ADL content, the biodegradation potential was calculated as the holocellulose/lignin ratio: (hemicellulose + cellulose)/lignin, with hemicellulose = NDF-ADF, and cellulose = ADF - ADL, expressed as percentage on OM content. Water soluble NO_3-N, Cl^-, SO_4^{2-} and Na^+ were measured after extraction (1:5 v/v water extraction ratio) according to EN 13652 with a Dionex DX-600 IC ion chromatograph (Dionex, Sunnyvale, CA). Water soluble NH_4-N was measured with a Skalar SAN++ flow analyzer (Skalar Analytical B.V, Breda, The Netherlands).

At the end of the composting process total Ca, K, Mg, Na, Fe, Mn, Al (< 10 mm fraction) and Cd, Cr, Cu, Pb, Ni, Zn and Mn (unsieved compost) concentrations were measured by a charge-coupled device (CCD) simultaneous Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES) (VISTA-PRO, Varian, Palo Alto, CA), after ashing and digestion with 7N HNO_3. Total P (< 10 mm fraction) was measured in the same extract with a Varian CARY 50 Spectrophotometer. Plant-available P, K, Ca, Mg, Fe, Mn, Cu and Zn (< 10 mm fraction) were measured in an ammonium acetate extract (1:5 v/v) at pH 4.65 and measured by a charge-coupled device (CCD) simultaneous Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES) (VISTA-PRO, Varian, Palo Alto, CA).

4.2.3. N immobilization

The separate bulking agents (on day 0) and the compost mixtures on day 14, 27, 55 and 76 (< 10 mm fraction) were tested for immobilization of mineral N by adding 350 mg N L^{-1} substrate as KNO_3 followed by 1 week of incubation at 37°C (Vandecasteele et al., 2016). Based on the difference between the theoretical (initial water-extractable mineral N concentration (EN 13652) + added amount of 350 mg N L^{-1} material) and actual water-

extractable mineral N content after this incubation period, the % N immobilization was calculated. As such, 100% N immobilization means that all of the 350 mg mineral N L^{-1} material is immobilized. By squeezing tightly a handful of the material (squeeze-ball test), it was determined if the material needed moistening before the incubation trial (when releasing at most only a few drops of water, the moisture content is sufficient).

4.2.4. Nitrogen fertilizer replacement value

A pot experiment with perennial ryegrass (*Lolium perenne* L., Melpetria tetra) started on January 27th 2015 in a greenhouse (average temperature of 20°C) with assimilation lights (from 5 am until 10 pm) to determine the nitrogen fertilizer replacement value (NFRV) of the four composts (products were stored at < 4°C until the pot trial was started). For comparison, also fresh leek residues were added. Additionally, N fertilizer was applied at four rates to the control soil (CON) to obtain N response curves necessary to calculate NFRV (CON_{0N}, CON_{50N}, CON_{100N}, CON_{150N}). Each treatment consisted of three replicates. The products were added based on an equal N input of 100 kg N ha^{-1}. P_2O_5 was added (as triple superphosphate) to adjust for an equal total P input of 47 kg ha^{-1} (the max. P input with Ct) in all treatments. To avoid K deficiency, additional KCl was supplied in the treatments with less than 100 kg K_2O ha^{-1} added with the products (more specifically 283 mg ground KCl to Ch). The pots (h = 15 cm, r = 11.3) were filled with enough soil to reach a bulk density of 1.4 g cm^{-3}. The pots were filled in two steps: half of the soil was added directly to the pot, after which the other half was filled with a mixture of soil, product and/or fertilizer. The pots were perforated at the bottom and placed on trays (height edge = 3 cm), to allow watering via the trays and to prevent nutrient leaching. On day 7, 50 seeds per pot were sown. The aboveground plant material was harvested on day 35 (first cut), day 61 (second cut), day 90 (third cut) and day 120 (fourth cut). The DM yield of the separate cuts was measured by drying the plant material for 48 h in a ventilated oven at 70°C. Afterwards, the total plant biomass from the four cuts per treatment and replicate were combined and ground (< 1 mm) in a plant mill (Fritsch pulverisette 19). The total N concentration of the grass was determined according to EN 13654-2, with a Thermo scientific-flash 4000 total N analyzer according to the Dumas method. To calculate the NFRV of the different products, first a crop response curve was plotted as a linear regression of the total plant N uptake in function of the mineral N fertilization rates in the control soil. N uptake was calculated based on the total N concentration and the DM crop yield of the grass. The NFRV is the N fertilization rate corresponding with the calculated plant N uptake of the treatment. Furthermore, at the

end of the experiment, soil samples were taken in each pot to determine mineral N (ISO 14256-2) to assess the N released from the added composts and crop residues, and pH-KCl (ISO 10390), to assess the effects of the amendments on soil pH.

4.2.5. Statistical analysis

A one-way ANOVA and post-hoc Scheffé test were used ($p < 0.05$) to assess the significant differences in product quality between the treatments at the same sampling time, and at the end of the NFRV experiment. Statistical tests were carried out using IBM SPSS Statistics for Windows, Version 21.0 (IBM Corp., Armonk, NY, USA). Further, we analyzed how the different treatments affected the product quality parameters (*Y*) throughout the experiment by taking the sampling time into account. Therefore, a linear regression model, including the effect of the *treatment*, the sampling time (*day*) and the interaction effect between the sampling time and treatment was fitted to the data: $Y = \beta_0 + \beta_{1,i} * Treatment + \beta_2 * Day + \beta_{3,i} * Day * Treatment + \epsilon$. In this model, the treatments were coded as categorical variables. Therefore, the effect of treatment 1 is included in the intercept (β_0), and hence, the coefficient of $\beta_{1,i}$ represents the effect of the other treatments ($i \in \{2, 3, 4\}$) relative to treatment 1. The effect of the sampling time (day) was included as a continuous variable, as such the coefficient β_2 gives the slope of the day effect for the reference treatment (treatment 1). Again, the coefficient of $\beta_{3,i}$ represents the change of the slope of the other treatments ($i \in \{2, 3, 4\}$) with respect to treatment 1. A normal distribution for the error term (ϵ) was specified with visual check of the residuals. The potential of N immobilization as a measure of compost stability was examined by comparing different regression models and Pearson correlation tests with the stability parameters OUR, biodegradation potential and C/N ratio. The open-source software platform R (version 3.1.3; R CoreTeam, 2015) was used for the regression models.

4.3. Results

4.3.1. Feedstock characterization: bulking agents, leek residues and mixtures

The two batches of leek residues added during the composting process had a very similar composition (Table 4.1). The BA were characterized by a higher OM and DM content, lower total N content and consequently higher C/N ratio than the leek residues (Table 4.1). In

comparison to wood chips and straw, the alternative BA had a lower OM content, higher N content and lower C/N ratio (Table 4.1). Chopped heath biomass had the lowest OM content (72.6% of DM), followed by tomato substrate (78.6% of DM) and strawberry substrate (88.2% of DM). Compared to the wood chips, total P content was lower in chopped heath and higher in the substrates. Especially tomato substrate had a high P concentration (6.8 g kg^{-1} DM). Water-soluble SO_4^{2-} and Cl^- concentrations were considerably higher in the substrates (SO_4^{2-}: 382 & 422 mg L^{-1} and Cl^-: 55 & 118 mg L^{-1} for the strawberry and tomato substrate, respectively) than in the chopped heath biomass and wood chips (SO_4^{2-}: < 11.7 mg L^{-1} and Cl^-: 17 & < 10 mg L^{-1}). NO_3^--N was high (401 mg L^{-1}) in the tomato substrate and negligible in the other BA. The spent growth media and wood bark had the lowest biodegradation potential (highest % lignin on OM content), followed by the chopped heath biomass (lowest % cellulose on the OM content), wood chips and straw (lowest % lignin and highest % hemicellulose and cellulose on the OM content). After mixing the separate feedstock materials, the initial Cw mixture had a higher C/N ratio (40.7), due to the higher C/N ratio of the wood chips, and a higher biodegradation potential (4.9) compared to the other mixtures (C/N between 25.7 and 32.8 and biodegradation potential between 2.5 and 3.0) (Appendix Table A4). The initial Ch mixture had a higher DM content (27%) than the initial Cs mixture (18%). No difference in fresh bulk density was observed among the four feedstock mixtures (Appendix Table A4).

4.3.2. Effect of bulking agents on composting process and compost quality

4.3.2.1. Pile temperature, CO_2 concentration and DM content

The average air temperature and total precipitation during the composting experiment were 7.1°C and 159 mm, respectively. Pile temperatures of Cw and Ch reached a maximum of 50-57°C after 5-8 days, respectively, while the temperature of Ct increased more slowly to 60°C after 15 days (Figure 4.1A). However, for Cw, temperatures started to decrease suddenly after 12 days to 14°C. In Cs temperatures did not exceed 42°C. The CO_2 concentrations (Figure 4.1B) in all piles, except Cw, were up to 20% in the beginning, indicating oxygen shortage. Consequently, the piles required frequent turning (12-13 times) and 200 kg straw was added to all piles after five days. Because the leek residues were already degraded (based on visual observations), extra fresh leek was added to all piles on day 19 to stimulate the microbial activity and enhance pile temperatures, especially in Cw and Cs.

Despite fresh leek addition, only a small temperature increase was noticed in Ch and Ct following fresh leek addition. CO_2 concentrations increased in Ch, Cs and Ct. Because CO_2 concentrations could not be sufficiently decreased in Ch, Cs and Ct and the pile temperatures of Cw were still low, it was decided to add an extra BA, i.e., poplar bark on day 28, which ameliorated the gas exchange in Ch, Cs and Ct and increased temperatures in all piles again. Especially pile temperatures of Cw increased strongly after day 28 (40-50°C). On day 34, all piles were compacted, which resulted in a similar temperature increase in all piles. Cw, Ch, Cs and Ct were turned 7, 10, 12 and 13 times in total, respectively.

At day 14, the DM content was lower for Cs and Ct compared to Cw and Ch, related to the initial wetter feedstock mixtures and the lower temperatures during composting (Table 4.1). With exception of Ct, the DM content of all other composts increased (Figure 4.2). At the end of composting, Cs still had a lower DM content (22.8% of fresh weight) compared to the other composts, while Ch was the driest compost (31.4%). There was no difference between the DM content of Cw and Ct (28.2 and 25.3%, respectively).

4.3.2.2. OM degradation: decomposition of OM and particle size distribution

For Cw and Ch, the OM content decreased (especially at the end of the composting process for Cw), while for Cs and Ct, the OM content not significantly changed compared to the beginning (Figure 4.2). There was a relative decrease of 23, 27, 4 and 8% compared to the start for Cw, Ch, Cs and Ct, respectively. During the entire composting period, Ch had the lowest OM content (47.9% of DM at the end).

On day 14, the fraction < 10 mm (Figure 4.3) was lower compared to the beginning in all mixtures (except in Cw), indicating that the smaller particles were degrading first. In Cw, the fraction < 10 mm had increased on day 14, indicating that the coarser particles were degrading first. The smaller particles in Cw started to decrease on day 27. Furthermore, on day 27, the fraction < 10 mm of Cs and Ch continued to decrease linearly, while in Ct it increased. From day 55 on, the fraction < 10 mm of all mixtures increased linearly, indicating that the coarser particles were decomposing or fragmenting. At day 76, Ch, Cs and Ct had a similar fraction < 10 mm (83-86%), while in Cw the fraction < 10 mm was lower (62%) and thus contained more coarse particles (confirmed by the presence of visible wood particles).

Table 4.1 Feedstock characterization of leek residues and bulking agents (mean ± standard deviation, n = 4). DM = dry matter, OM = organic matter, nda = no data available. Parameters of bulking agents with the same letter are not significantly different from each other (Scheffé test, $p < 0.05$).

	Crop residues		Bulking agents											
	Leek	Leek, extra	Poplar wood chips		Chopped heath		Strawberry substrate		Tomato substrate		Rye straw		Poplar wood bark	
Fresh bulk density (kg m^{-3})	332 ± 161	186 ± 18	375		266		374		205		12		278 ± 19	
Organic matter content (% of DM)	54.3 ± 11.7	57.4 ± 11.5	94.3 ± 0.7	d	72.6 ± 4.3	a	88.2 ± 0.9	c	78.6 ± 1.7	b	96.1 ± 0.4	d	88.7 ± 0.2	c
Dry matter content (% of fresh weight)	13.2 ± 1.9	13.0 ± 2.3	26.8 ± 0.4	a	35.3 ± 0.1	a	19.2 ± 0.4	a	28.1 ± 0.5	a	64.0 ± 12.4	a	37.3 ± 2.8	a
C/N (-)	15.6 ± 2.7	17.4 ± 1.2	92.5 ± 1.6	c	39.3 ± 6.5	ab	35.5 ± 1.6	a	27.4 ± 1.0	a	118.4 ± 10.1	d	51.1 ± 5.2	b
C/P (-)	110.1 ± 14.2	117.6 ± 9.4	690.5 ± 21.4	a	798.5 ± 67.9	a	270.8 ± 8.1	a	65.0 ± 7.7	a	465.4 ± 140.8	a	526.9 ± 64.5	a
Total N (g kg^{-1} DM)	19.39 ± 3.34	18.50 ± 4.26	5.67 ± 0.13	a	10.40 ± 1.08	b	13.82 ± 0.64	c	15.98 ± 0.56	d	4.54 ± 0.36	a	9.72 ± 0.97	b
Total P (g kg^{-1} DM)	2.76 ± 0.59	2.72 ± 0.58	0.76 ± 0.02	b	0.51 ± 0.03	a	1.81 ± 0.04	d	6.79 ± 0.75	e	1.22 ± 0.33	cd	0.95 ± 0.13	bc
Total Cd (mg kg^{-1} DM)	0.17 ± 0.03	0.23 ± 0.04	0.80 ± 0.02	d	0.27 ± 0.03	c	0.14 ± 0.02	b	0.15 ± 0.01	b	0.05 ± 0.01	a	1.67 ± 0.07	e
Total Cr (mg kg^{-1} DM)	3.33 ± 1.08	5.18 ± 1.35	1.05 ± 0.72	b	2.30 ± 0.43	c	2.10 ± 0.54	bc	17.70 ± 6.31	d	0.20 ± 0.00	a	1.90 ± 0.63	bc
Total Cu (mg kg^{-1} DM)	7.55 ± 0.60	8.61 ± 1.64	3.88 ± 0.19	b	7.47 ± 0.65	d	16.37 ± 0.95	e	54.25 ± 1.48	f	2.31 ± 0.24	a	5.31 ± 0.18	c
Total Pb (mg kg^{-1} DM)	14.28 ± 2.69	17.50 ± 5.85	1.30 ± 0.22	ab	17.18 ± 2.64	d	3.98 ± 0.43	bc	4.50 ± 0.50	c	0.73 ± 0.10	a	5.73 ± 0.29	c
Total Ni (mg kg^{-1} DM)	1.38 ± 0.46	2.58 ± 0.62	0.48 ± 0.05	a	3.00 ± 0.36	c	1.55 ± 0.10	b	6.00 ± 0.63	d	0.25 ± 0.06	a	1.45 ± 0.10	b
Total Zn (mg kg^{-1} DM)	28.68 ± 2.92	35.80 ± 5.79	55.88 ± 2.67	b	60.08 ± 6.94	bc	73.13 ± 4.13	c	88.55 ± 4.73	d	15.83 ± 4.94	a	194.15 ± 5.72	e
Total Mn (mg kg^{-1} DM)	84.63 ± 14.56	70.20 ± 7.27	37.83 ± 14.66	b	140.78 ± 9.03	d	116.45 ± 13.55	cd	1325.95 ± 311.06	e	11.65 ± 0.90	a	79.00 ± 7.41	c
Total K (g kg^{-1} DM)	23.60 ± 3.98	22.32 ± 3.60	3.28 ± 0.06	a	1.41 ± 0.08	a	7.13 ± 4.09	a	9.51 ± 0.22	a	5.27 ± 0.29	a	5.64 ± 1.27	a
Total Mg (g kg^{-1} DM)	1.63 ± 0.22	1.56 ± 0.07	0.58 ± 0.01	a	0.54 ± 0.03	a	3.62 ± 0.18	a	5.80 ± 0.31	a	0.36 ± 0.04	a	2.00 ± 0.11	a
Total Ca (g kg^{-1} DM)	9.36 ± 1.88	9.91 ± 1.75	6.87 ± 0.24	b	3.08 ± 0.33	a	20.92 ± 1.06	c	34.97 ± 2.16	e	2.13 ± 0.24	a	27.11 ± 1.17	d
Total Na (g kg^{-1} DM)	0.33 ± 0.05	0.37 ± 0.05	0.10 ± 0.00	b	0.11 ± 0.01	b	0.50 ± 0.02	d	1.60 ± 0.09	e	0.05 ± 0.01	a	0.17 ± 0.01	c
pH-H_2O (-)	nda	nda	6.88 ± 0.07	c	5.95 ± 0.05	a	6.68 ± 0.14	bc	6.45 ± 0.04	abc	6.14 ± 0.50	bc	7.48 ± 0.10	d
Electrical conductivity (µS cm^{-1})	nda	nda	104 ± 11	b	48 ± 4	a	461 ± 46	d	978 ± 131	d	186 ± 64	c	169 ± 15	c
NO_3^--N (mg L^{-1})	nda	nda	< 5.0		< 5.0		< 5.0		401 ± 64		< 5.0		1 ± 3	
NH_4^+-N (mg L^{-1})	nda	nda	< 5.0		< 5.0		9 ± 5		8 ± 1		8 ± 10		< 5.0	
SO_4^{2-} (mg L^{-1})	nda	nda	< 11.7		< 11.7		382 ± 14		422 ± 74		27 ± 19		19 ± 18	
Cl^- (mg L^{-1})	nda	nda	< 10.0		17 ± 1		55 ± 2		118 ± 18		98 ± 38		22 ± 10	
Hemicellulose (% of OM)	11.7 ± 10.4	9.2 ± 11.1	22.5 ± 2.1	a	32.6 ± 6.2	bc	21.2 ± 2.4	a	28.5 ± 1.7	b	33.6 ± 1.9	c	21.7 ± 4.2	a
Cellulose (% of OM)	23.3 ± 6.3	24.7 ± 7.2	48.9 ± 1.2	c	19.9 ± 3.8	a	28.8 ± 1.7	b	26.2 ± 2.2	b	49.5 ± 1.5	c	29.5 ± 3.1	b
Lignin (% of OM)	4.5 ± 1.3	4.2 ± 0.8	19.5 ± 0.3	b	24.2 ± 1.5	c	30.8 ± 1.0	d	29.8 ± 1.6	d	8.5 ± 0.1	a	28.5 ± 1.8	d
Biodegradation potential (-)	7.9 ± 3.6	8.0 ± 3.5	3.7 ± 0.1	c	2.2 ± 0.3	b	1.6 ± 0.1	a	1.8 ± 0.1	ab	9.7 ± 0.3	d	1.8 ± 0.2	a

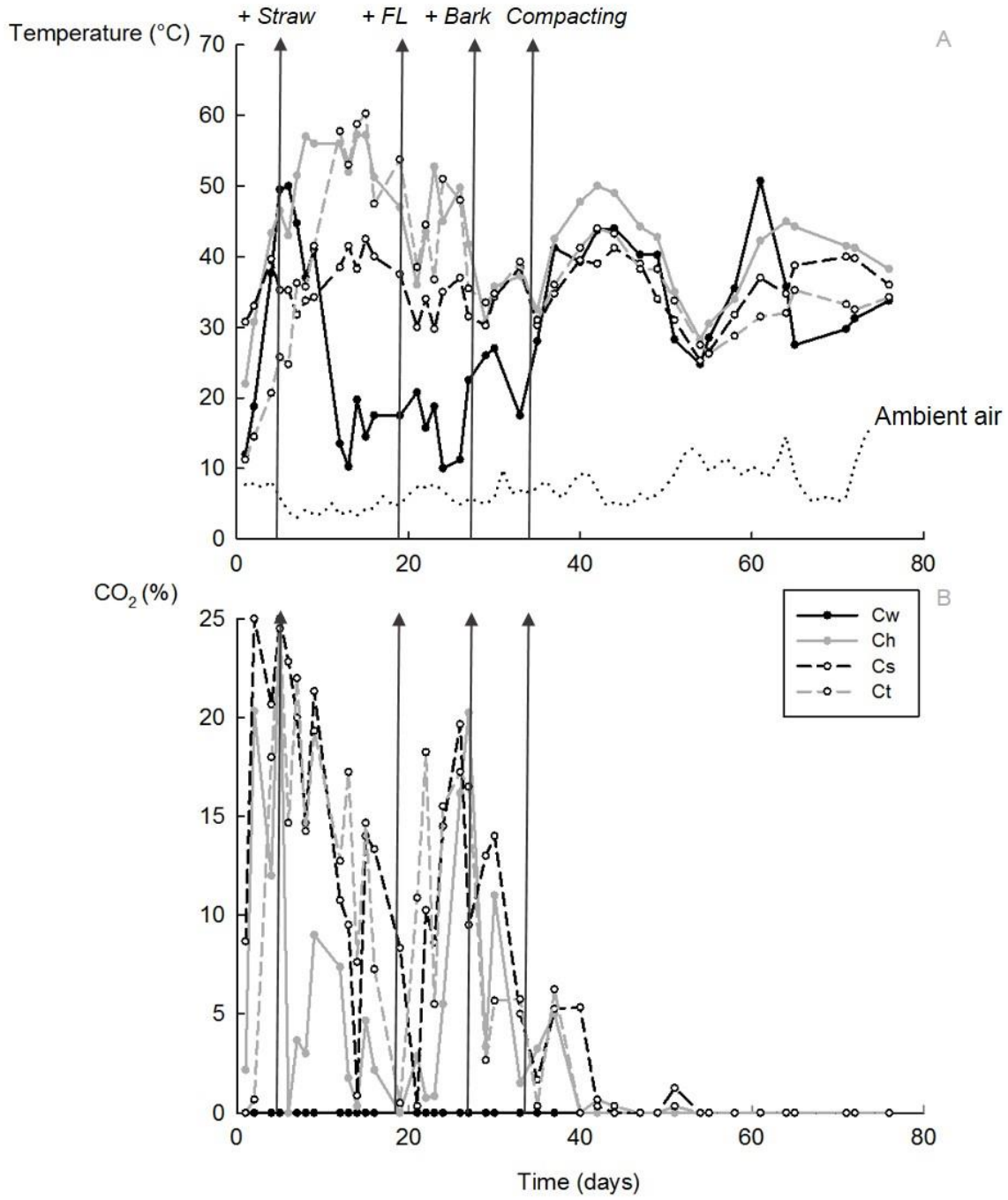

Figure 4.1 Temperature (A) and CO_2 (B) profiles during co-composting of leek residues with wood chips (Cw), chopped heath biomass (Ch), spent strawberry (Cs) or tomato (Ct) substrate (mean of 4 measurements, standard deviations were below 10°C and 9% CO_2). On day 5, 19 and 28, respectively, straw, fresh leek (FL) and wood bark were added, on day 34 the piles were compacted.

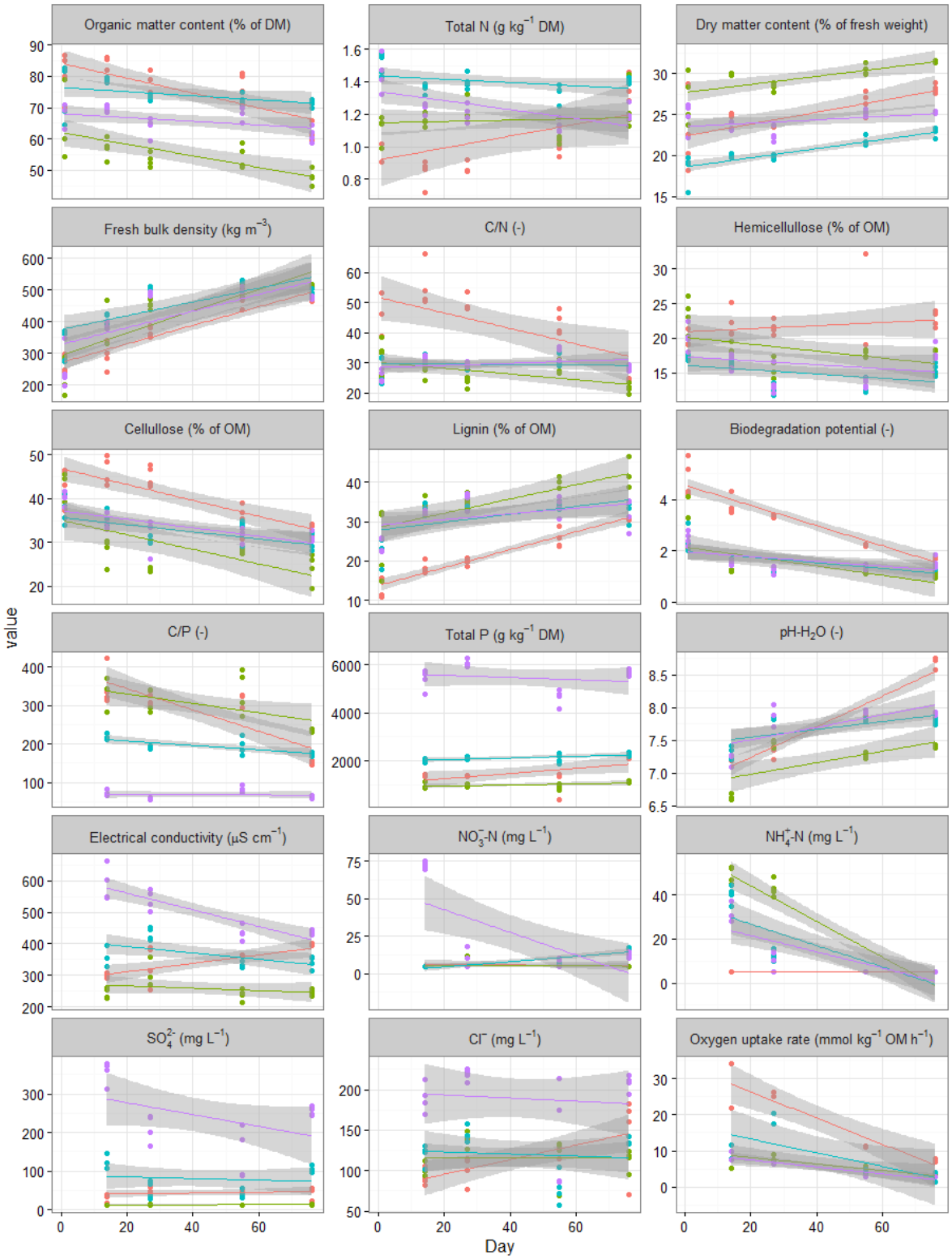

Figure 4.2 Output of the regression model for the different quality parameters of the compost mixtures at day 0, 14, 27, 55 and 76 of co-composting leek residues with wood chips (red), chopped heath biomass (green), spent strawberry (blue) or tomato substrate (purple) (n = 4). DM = dry matter, OM = organic matter. The grey shades indicate 0.95 confidence intervals. Regression coefficients can be found in Appendix Table A5.

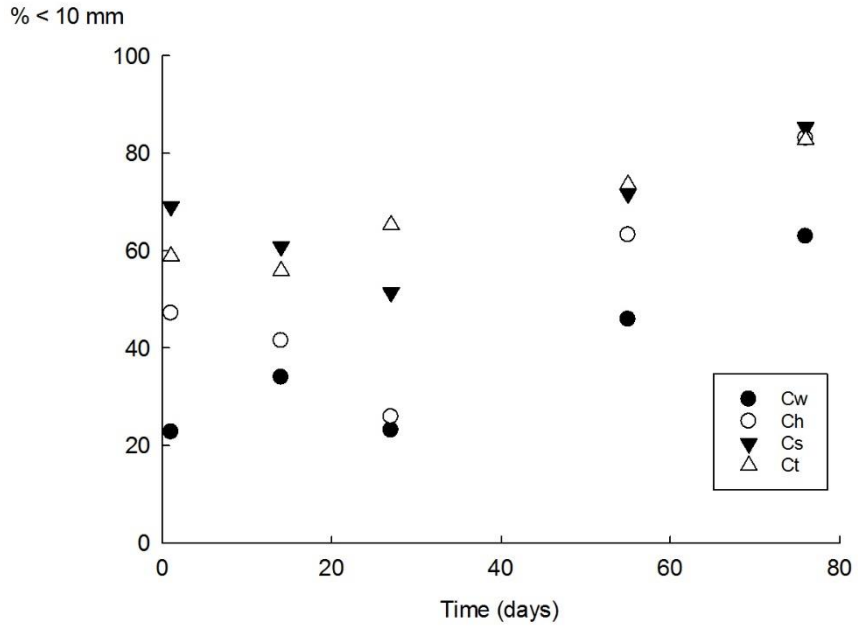

Figure 4.3 Percentage of fraction < 10 mm in compost mixtures at day 0, 14, 27, 55 and 76 of co-composting leek residues with wood chips (Cw), chopped heath biomass (Ch), spent strawberry (Cs) or tomato (Ct) substrate.

4.3.3. Compost stability

Compost stability was assessed by a set of individual parameters, i.e., C/N ratio, mineral N (NO_3^--N and NH_4^+-N) concentrations and their ratio, OUR, biodegradation potential and N immobilization. Parallel to the OM content, the C/N ratio decreased for Cw and Ch, but hardly changed for Cs and Ct during the composting process (Figure 4.2). For Cw, the C/N ratio was higher compared to the other mixtures at the start, but showed a steeper decrease (Figure 4.2). Hence, C/N ratios were similar at the end of the composting process for all treatments. The total mineral N concentration was low (< 5 mg L^{-1}) for Cw during the whole composting period. For Ch the NH_4^+-N concentration decreased after 27 days and no considerable NO_3^--N was measured. In contrast, for Cs and Ct the NH_4^+-N concentration had already decreased after 14 days along with an increase in NO_3^--N and a NO_3^--N/NH_4^+-N, indicating stabilization (Appendix Table A4). For all treatments, the OUR decreased during the composting process, with the strongest decline for Cw (Figure 4.2). At day 14, the OUR of Ch, Cs and Ct had typical values of stable composts (5-10 mmol kg^{-1} OM h^{-1} according to Flemish compost standards), indicating that the material is little biodegradable, while the OUR of Cw (28.0 mmol kg^{-1} OM h^{-1}) was much higher and hence Cw was not stabilized yet (Appendix Table A4). The relative proportion of hemicellulose (on OM) did not change

during the composting process for the different treatments (Figure 4.2). The relative proportion of cellulose decreased for all treatments (Figure 4.2), with the strongest decrease for Cw and Ch. The relative proportion of lignin increased for all treatments and was most pronounced for Cw and Ch. For Ch, Cs and Ct, the strongest increase occurred from the start until day 14, while for Cw the strongest increase was during the last month (Appendix Table A4). The relative proportions of cellulose and hemicellulose were higher and the relative proportion of lignin was lower for Cw compared to the other mixtures on day 14, 27 and 55 (Appendix Table A4). This resulted in a higher biodegradation potential for Cw. Ch was generally characterized by a lower relative cellulose and higher lignin proportion compared to the other mixtures, resulting in a lower biodegradation potential. The biodegradation potential of all composts had decreased compared to the beginning, with the strongest decrease for Cw (Figure 4.2).

At the end of the composting process, the C/N ratio of Ch (21.6) was lower compared to the other composts (between 25.2 and 28.4). Further, Cw could be considered stable and Ch, Cs and Ct could be considered very stable composts according to the OUR (< 5 mmol kg^{-1} OM h^{-1}) (Appendix Table A4). This was confirmed by the NO_3^--N/NH_4^+-N ratio > 1 for Cs and Ct and the biodegradation potential < 1.8 for all composts (Blanco and Almendros, 1997; Eklind and Kirchmann, 2000; Francou et al., 2008; Lashermes et al., 2012; Vandecasteele et al., 2014).

Clear differences in N immobilization of added mineral N among treatments during the composting process and at the end were observed (Figure 4.4). N immobilization in the compost mixtures at day 14 was higher compared to the raw BA at the start of the compost experiment, partially explained by adding rye straw and poplar bark. N immobilization in Cw was high, on average 92 ± 11% and did not change during the composting process. For Ch and Cs, N immobilization was high until day 55 (on average 95 ± 6% and 73 ± 24%, respectively), while on day 76 it decreased to 30 ± 13% for Ch. N immobilization was lower in Ct (on average 22 ± 28% until day 55). On day 76, N immobilization in Cs and Ct was negligible (-3 ± 0% and -2 ± 12%). These negative values indicate low N release during the incubation trial. The two composts with low mineral N content at the end of the process thus also had the highest N immobilization.

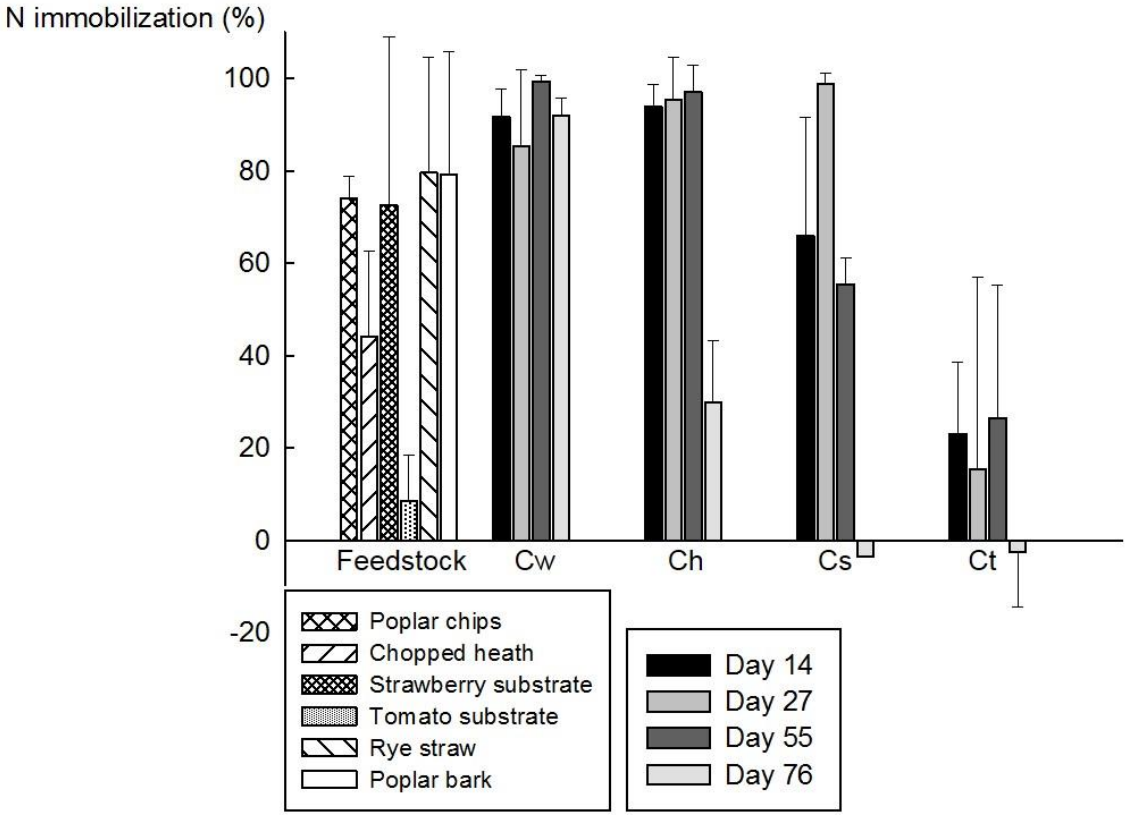

Figure 4.4 Percentage N immobilization in separate feedstock materials and in compost mixtures at day 14, 27, 55 and 76 of co-composting leek residues with wood chips (Cw), chopped heath biomass (Ch), spent strawberry (Cs) or tomato (Ct) substrate. Error balks indicate standard deviations (n = 4).

4.3.4. Compost characteristics

4.3.4.1. Fresh bulk density and pH

Generally, the fresh bulk density increased as the composting process evolved, especially during the first month (Figure 4.2). Cw was characterized by a lower fresh bulk density than the other mixtures until day 55, and a lower fresh bulk density than Ch on day 76. For Cw, the pH continued to increase, while for the other mixtures the initial pH increase stagnated after 27 days (Appendix Table A4), resulting in the highest pH for Cw at the end of the composting process (8.69). pH was lower for Ch compared to the other mixtures during the whole composting process (7.41 at day 76), related to the more acidic character of the chopped heath biomass (Table 4.1). In the pot trial with perennial ryegrass, a higher soil pH-KCl was observed in soils with compost added (except for Ch) in comparison to the control soil (no compost addition) (Table 4.2).

Table 4.2 DM crop yield, N content in the crop (%) and N uptake, nitrogen fertilizer replacement value (NFRV) and soil pH-KCl at the end of the perennial ryegrass pot trial, with amendment of fresh leek residues (FL) and co-composted leek residues with wood chips (Cw), chopped heath biomass (Ch), spent strawberry (Cs) or tomato (Ct) substrate (mean ± standard deviation, $n = 4$) at an N input of 100 kg N ha^{-1}. Mineral N fertilizer was applied at four rates to the control soil without amendment (CON). Parameters with the same letter are not significantly different from each other (Scheffé test, $p < 0.05$).

	DM crop yield (kg ha^{-1})		% N in crop (of DM)		N uptake (kg ha^{-1})		NFRV (kg kg^{-1} N)		pH-KCl (-)	
CON	2107 ± 189	a	1.62 ± 0.05	a	31 ± 3	a	-		6.11 ± 0.09	bc
CON_{50N}	3597 ± 157	b	2.02 ± 0.10	abc	67 ± 6	b	-		6.07 ± 0.03	bc
CON_{100N}	4961 ± 66	c	2.13 ± 0.18	bc	97 ± 7	c	-		5.89 ± 0.05	ab
CON_{150N}	5771 ± 106	c	2.35 ± 0.21	c	125 ± 12	d	-		5.71 ± 0.08	a
FL	3401 ± 149	b	1.94 ± 0.03	ab	61 ± 3	b	0.44 ± 0.05	b	6.18 ± 0.03	cd
Cw	2271 ± 76	a	1.61 ± 0.03	a	34 ± 2	a	0.01 ± 0.03	a	6.44 ± 0.04	e
Ch	2236 ± 108	a	1.67 ± 0.11	a	34 ± 2	a	0.02 ± 0.03	a	6.31 ± 0.11	cde
Cs	2079 ± 195	a	1.68 ± 0.12	a	32 ± 1	a	-0.02 ± 0.02	a	6.48 ± 0.08	e
Ct	2284 ± 74	a	1.65 ± 0.04	a	35 ± 1	a	0.03 ± 0.02	a	6.42 ± 0.07	de

4.3.4.2. Total nutrient contents

Cw and Cs were, respectively, characterized by the lowest and highest N content during the composting process (Appendix Table A4), which reflected the initial differences in the raw BA (wood chips versus strawberry substrate, Table 4.1). At the end, however, no difference in total N content was observed among the four composts (between 12 and 14 g kg^{-1} DM) as the N content of Cw increased during the composting process (Figure 4.2). Total P content increased for Cw from day 14 until 76, but did not change for Ch, Cs and Ct (Appendix Table A4). Ch was characterized by the lowest P content (1.14 g kg^{-1} DM) and highest C/P ratio (232), and Ct by the highest P content (5.69 g kg^{-1} DM) and lowest C/P ratio (60) at the end of the composting process (Appendix Table A4). Furthermore, the end product of Ch was also characterized by the lowest Mg, K, Na and Zn content and highest Pb content (Table 4.3). Cw had a higher Cd content, and Ct was characterized by a higher Cu, Cr, Na and Mn content as compared to the other composts (Table 4.3). Heavy metals (Cd, Cr, Cu, Pb, Ni and Zn) did not exceed the maximum permitted range for fertilizers or soil improvers in Flanders (Table 4.3). However, both the wood chips and the poplar bark are an important source of Cd, and to a lesser extent of Zn in the composts, as reported previously (Vandecasteele et al., 2013).

Table 4.3 Plant-available (extracted in ammonium acetate (AmAc)) and total nutrient concentrations in the end products of co-composted leek residues with wood chips (Cw), chopped heath biomass (Ch), spent strawberry (Cs) or tomato (Ct) substrate (mean ± standard deviation, $n = 4$). DM = dry matter. Parameters with the same letter are not significantly different from each other (Scheffé test, $p < 0.05$). As a reference, the optimal ranges for plant-available nutrients for use in potting soil (De Tender et al., 2016) and the Flemish heavy metal limits for use as soil improvers and fertilizers (Flemish regulation concerning the sustainable management of materials and waste) are given.

	Cw		Ch		Cs		Ct		Optimal range potting soil/Flemish heavy metal limit soil improvers/fertilizers
P-AmAc (mg L^{-1} substrate)	143 ± 5	b	89 ± 2	a	170 ± 3	b	551 ± 26	c	30 - 70
K-AmAc (mg L^{-1} substrate)	1206 ± 25	c	867 ± 32	a	1122 ± 33	b	1228 ± 21	c	150 - 360
Ca-AmAc (mg L^{-1} substrate)	1356 ± 48	b	1134 ± 29	a	1952 ± 32	c	2691 ± 73	d	325 - 2100
Mg-AmAc (mg L^{-1} substrate)	182 ± 7	a	181 ± 3	a	324 ± 10	b	412 ± 9	c	150 - 300
Mn-AmAc (mg L^{-1} substrate)	7.4 ± 0.3	b	4.5 ± 0.2	a	8.7 ± 0.8	b	29 ± 2	c	
Zn-AmAc (mg L^{-1} substrate)	4.8 ± 0.1	c	4.5 ± 0.1	bc	3.7 ± 0.2	a	4.3 ± 0.1	b	0.5 - 10
Na (mg L^{-1} substrate)	43.5 ± 1.2	a	41.1 ± 0.8	a	56.7 ± 1.7	b	169.7 ± 4.7	c	
Total Mn (g kg^{-1} DM)	0.1 ± 0.0	a	0.2 ± 0.0	b	0.2 ± 0.0	b	1.2 ± 0.2	c	
Total K (g kg^{-1} DM)	10.3 ± 0.3	c	5.9 ± 0.4	a	10.7 ± 0.2	bc	10.5 ± 0.4	b	
Total Mg (g kg^{-1} DM)	1.9 ± 0.1	a	1.4 ± 0.0	b	3.5 ± 0.1	c	4.7 ± 0.1	d	
Total Ca (g kg^{-1} DM)	16.4 ± 1.0	a	8.8 ± 0.5	a	22.8 ± 0.8	a	33.1 ± 2.1	a	
Total Na (g kg^{-1} DM)	0.2 ± 0.0	b	0.2 ± 0.0	a	0.4 ± 0.0	c	1.0 ± 0.0	d	
Total Fe (g kg^{-1} DM)	2.1 ± 0.3	a	2.1 ± 0.1	a	2.1 ± 0.1	a	5.2 ± 1.1	b	
Total Al (g kg^{-1} DM)	1.4 ± 0.1	a	1.4 ± 0.1	a	1.6 ± 0.0	a	1.8 ± 0.1	a	
Total Cd (mg kg^{-1} DM)	1.1 ± 0.0	c	0.5 ± 0.0	ab	0.5 ± 0.0	a	0.6 ± 0.0	b	2
Total Cr (mg kg^{-1} DM)	3.8 ± 0.5	a	4.9 ± 0.7	a	3.5 ± 0.2	a	21.4 ± 16.3	b	70
Total Cu (mg kg^{-1} DM)	9.1 ± 0.2	a	10.7 ± 1.3	a	15.2 ± 0.5	b	38.8 ± 1.9	c	150
Total Pb (mg kg^{-1} DM)	10.8 ± 0.6	a	22.5 ± 1.6	b	9.2 ± 0.3	a	10.8 ± 1.3	a	150
Total Ni (mg kg^{-1} DM)	3.4 ± 1.7	a	4.3 ± 1.4	a	2.4 ± 0.5	a	4.7 ± 0.4	a	30
Total Zn (mg kg^{-1} DM)	106 ± 2	b	92 ± 5	a	103.9 ± 1.7	b	110 ± 5	b	400

4.3.4.3. Plant-available nutrients and EC

Water-soluble Cl^- concentrations did not change during the composting process for Ch, Cs and Ct (Figure 4.2), and were not different for the mixtures with alternative BA compared to Cw at the end (Appendix Table A4). Water-soluble SO_4^{2-} concentrations of the end products were lowest for Ch, followed by Cw and did not change during the composting process for Ch and Cw (Figure 4.2). For Ct, the SO_4^{2-} concentration showed an overall decreasing trend (Figure 4.2). The SO_4^{2-} and Na^+ concentrations of the end products were higher for Cs than for Ch and Cw, but were the highest for Ct (more than four times higher than Cw). EC increased only for Cw and did not change for the mixtures with alternative BA (Figure 4.2). EC was lowest for Ch and highest for Ct during the whole process, related to the higher concentrations of soluble nutrients. Plant-available nutrients in the end products (P, Ca, Mg, Mn) were all highest for Ct and lowest for Ch. Plant-available K was equally highest for Cw and Ct (Table 4.3).

4.3.4.4. Nitrogen Fertilizer Replacement Value

We observed a linear response of crop N uptake to total N supply (N uptake = 0.62 * N supply + 33.11, R^2 = 0.97, p < 0.01) for the reference treatments with mineral N supply. Control soil at zero N fertilizer rate released a significant amount of N, as indicated by the crop uptake of 31 kg ha^{-1}. There was no effect of compost type on DM crop yield, % N in the crop, N uptake and NFRV (Table 4.2). All composts had a DM crop yield comparable to the control treatment at zero N fertilizer rate and an NFRV around 0, in contrast to the fresh leek residues with a NFRV of 0.44 kg kg^{-1} N and a significantly higher DM crop yield and N uptake than the control at zero N fertilizer rate and the composts, indicating considerably higher N mineralization rates from crop residues. Furthermore, we observed low soil mineral N concentrations at the end of the experiment (< 4 kg ha^{-1}), which points to N as the limiting element for plant growth.

4.4. Discussion

4.4.1. Composting process: temperature and CO_2 patterns

After a first temperature peak, pile temperatures of Cw suddenly decreased to 14°C after 12 days, while enough oxygen was available. Five hypotheses could explain why the composting process slowed down for Cw: (1) A lack of easily available N. This seems unlikely, as adding extra N via 2250 kg fresh leek on day 19 did not cause pile temperatures to rise, consequently N is certainly not limiting (Figure 4.1A). (2) A lack of easily available C. Because almost all N became immobilized after adding extra N in the N immobilization experiment (Figure 4.4) and no significant temperature rise was noticed after adding extra available C with the wood bark on day 28 (Figure 4.1A), it can be assumed that enough C was available. (3) A lack of oxygen. According to the low CO_2 measurements (Figure 4.1B) and low NH_4^+-N concentrations in relation to the NO_3^--N concentrations in the mixture (Appendix Table A5), sufficient oxygen is available and this hypothesis was also rejected. (4) Inhibition of the microbial activity due to toxic compounds as volatile fatty acids were released during the degradation process. This is assumed to be unlikely as there was no microbial inhibition during the initial phase of the OUR measurement (data not shown). (5) Cooling down of the compost pile due to excessive porosity of the mixture caused by the large particle size of the wood chips. Excessive porosity would result in too much aeration, causing rapid dissipation of heat. This hypothesis was considered to be most likely as the

addition of a denser BA (wood bark) on day 28 and consequently turning the pile, thereby making it more compacter, caused a slight increase in pile temperature (Figure 4.1A). Even more convincing is the fact that when the pile was compacted on day 34, which thus decreased the porosity of the mixture, the pile temperatures increased considerably (Figure 4.1A). Additionally, the OM loss, the decrease in the share of cellulose and increase in the share of lignin and pH (degradation of organic acids) all indicate increased microbial activity and degradation of Cw during the second month.

Compared to wood chips, spent strawberry and tomato substrates led to lower pile temperatures (max. 42°C) during the first nine days of composting (before the temperature drop in Cw). It is supposed that this was a pure physical effect due to the pores being blocked by the dense structure of the substrates. Aeration was lower, leading to anaerobic conditions (as confirmed by the high CO_2 concentrations) that hindered the growth of aerobic microorganisms and thus the composting process (Alburquerque et al., 2006). Turning the pile and adding a coarser BA (wood bark) reduced the density of the mixture, resulting in lower CO_2 concentrations and thus increased oxygen availability. Additionally, the initial substrates and mixtures at the start were less biodegradable (lower OUR and biodegradation potential) compared to the wood chips and chopped heath biomass and their mixtures. This is related to the recalcitrant C in the peat (the major component of the substrates). Only a small decrease in OM content and C/N ratio during the process were measured in Cs and Ct in contrast to a similar decrease in Cw and Ch, despite the increased pile temperature of Ct after 12 days. A possible reason for the lower temperature development and OM decomposition in Cs and Ct may be related to slower decomposition of peat and the fast microbial decomposition of the strawberry and tomato roots, resulting in a faster depletion of easily degradable C in the substrates. OM losses of Ch were slightly higher (27%) than Cw (23%), related to the highest pile temperatures and highest level of microbial activity in Ch. This confirmed that the low pH of Ch, related to the low pH of the chopped heath biomass, did not have a negative effect on the microbial community. Furthermore, the large amount of soluble phenolic compounds and the related recalcitrant humic complexes in the chopped heath biomass (Kristensen and McCarty, 1999; Kuiters, 1990) did not limit OM degradation. Nevertheless, temperatures did not surpass the required temperatures in Belgium to ensure compost sanitation, i.e., 60°C for a continuous period of at least 4 days or 55°C for a continuous period of at least 12 days, in any of the piles. A more balanced feedstock mixture with more easily degradable C could help to achieve higher temperatures.

The lower temperatures resulted in less evaporation, explaining the low DM content (between 22.8-31.4%) of the composts (Belgian federal norm is 50%).

4.4.2. Compost quality

All BA resulted in composts with an OM content suitable for use as soil amendment, with high values of lignin (31-40% of OM), resistant to microbial decomposition and consequently contributing to the buildup of soil organic carbon (SOC) (Foston and Ragauskas, 2012). The higher share of lignin and decreased biodegradation potential in combination with the decrease in OUR as the composting process proceeded indicated increased stabilization of all composts. Additionally, lower N release and thus slower decomposition was noticed after compost application in comparison with the fresh leek residues, another indicator of compost stability. Using NO_3^--N/NH_4^+-N ratios > 1 and the absence of N immobilization as evaluation criteria, Ct and Cs could be considered most stable as they had considerable NO_3^--N concentrations in contrast to Cw and Ch. For Ct, however, this was related to the initially high NO_3^--N in the tomato substrate (due to fertilization during the cultivation of tomatoes), which also explained the lower N immobilization rates during the composting process compared to the other composts. For Cs, a clear conversion from NH_4^+-N to NO_3^--N (indicating more aerobic conditions in the compost) during the last 20 days of composting explained the increased NO_3^--N/NH_4^+-N ratio and decreased N immobilization at day 76. Those high NO_3^--N concentrations also explained the lower pH of Cs and Ct, as the conversion from NH_4^+-N to NO_3^--N results in acidification of the composts (Sánchez-Monedero et al., 2001). Cw and Ch had low mineral N concentrations and immobilized more N during the entire composting process. Cw was less stable and its initial OM was less decomposed compared to the other composts as indicated by the higher OUR and biodegradation potential, more coarser particles and the highest N immobilization of all the composts. Vandecasteele et al. (2016) reported that higher microbial activity in green waste composts (indicated by higher oxygen consumption) and thus a lower compost stability resulted in higher N immobilization in the compost. Despite the differences in mineral N content among the composts, there was no effect on the DM crop yield and N uptake of perennial ryegrass. In addition, the soil pH was higher at the end of the pot trial when adding Cw, Cs and Ct compared to no compost addition.

4.4.3. Practical implications

The choice of the BA is dependent on the compost requirements and availability of the BA. Depending on the initial characteristics of the BA, the final compost had lower (Ch) or higher (Cs and Ct) total and plant-available nutrient contents, which should be taken into account during field application. The C/P ratio of organic fertilizers is gaining importance due to environmental concerns related to P leaching losses from agricultural fields, resulting in restrictions for P input (Vanden Nest et al., 2016). The C/P ratio determines the amount of C added to the soil per unit of P. Especially Ch seems to be suited for improving SOC in soils already rich in P, given its high C/P ratio. In contrast, Ct was the most nutrient-rich compost with the highest EC, but no problems with excessive salt concentrations are expected (EC < 750 µS cm^{-1} for use in growth media). Furthermore, because Ct had the lowest C/P ratio, a lower application dose is recommended for soils rich in P. An excess of nutrients in the growth media during cultivation may interfere with the reuse of growth media as BA for composting. Using the alternative BA resulted in less Cd in the compost compared to using wood chips, which could help to reduce the Cd input in agricultural soils. Because peat extraction and export have a severe environmental impact (Cleary et al., 2005), we assessed the composts for use in growth media. When comparing the pH, EC and plant-available nutrients with optimal ranges for use in potting soil according to De Tender et al. (2016) (pH: 4.5-7.0 and EC: 200-750 µS cm^{-1}, nutrients: Table 4.3), it seemed that Ch has the most potential to be used as peat substitute.

The availability of the BA is determined by the total available amounts, seasonal and geographic availability. In our case, the spent tomato substrates were only available in December, the spent strawberry substrates between March and December and the chopped heath biomass between August and February. Regarding the total available amounts and seasonal availability, spent strawberry substrate and chopped heath biomass have the most potential in Flanders (see Chapter 8, Table 8.1). However, a mixture of different types of BA resulted in a better composting process, as such, the choice of the BA is especially dependent on local opportunities.

4.5. Conclusions

The aim of this research was to test alternative bulking agents (BA), i.e., chopped heath biomass and spent growth media used in cultivation of strawberry and tomato, as alternatives for wood chips when co-composting leek residues. We conclude that chopped heath biomass and both types of spent growth media can be used as BA for co-composting green crop residues, but they resulted in different composting processes and end products. More specifically, when using spent growth substrates, the compost requires more frequent turning in comparison with wood chips due to oxygen shortage from compaction of the mixtures. To prevent oxygen shortage when applying these alternative BA, a coarser BA (i.e., larger particle size) with more structure could be added. Due to the suboptimal feedstock mixtures, pile temperatures did not surpass the required temperatures to ensure compost sanitation. Use of spent growth media resulted in lower OM degradation compared to chopped heath biomass, because the substrates had a lower biodegradation potential related to the high content of recalcitrant C in the peat. Nevertheless, all BA resulted in composts with a high OM content and thus a higher potential for enhancing SOC in agricultural soils. Especially chopped heath biomass has potential for (1) improving C content in P-rich soils due to its high C/P ratio and (2) for peat replacement in growth media based on pH, EC and plant available nutrients. Depending on the initial nutrient content of the BA, the final compost had lower (chopped heath biomass) or higher (spent growth media) nutrient contents. Avoiding excess nutrient application during soilless cultivation could decrease nutrient contents of the spent growth media and would consequently result in a compost with a lower nutrient content and EC. The nutrient content should be taken into account during field application, hence, context-specific soil and crop requirements and local availability (to reduce transport costs) could help determining the most appropriate BA and BA mixture.

CHAPTER 5

Field storage conditions for cattle farmyard manure

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Abstract

Storage and application of cattle farmyard manure (CFM) can cause considerable environmental problems through nutrient losses to soil, water and air, if not properly handled. We investigated different storage conditions of CFM at field scale to reduce nitrogen (N) losses to the soil, meanwhile optimizing the agronomical quality of the CFM. The treatments differed in terms of storage method (stockpiling, extensive composting or co-composting with bulking agents) and coverage (no cover, plastic or geotextile cover). Over the different treatments, the ammonium-N concentrations under the piles in the 0–90 cm soil layer amounted to a maximum of 4.2% of the initial manure N content. We were able to assess the relative importance of each of the two processes resulting in a higher mineral N concentration under the piles, i.e. direct leaching from the CFM to the soil on the one hand, and a smaller indirect effect of elevated soil temperatures (up to 37°C) under the piles resulting in higher N mineralization in the top soil on the other hand. NH_4^+-N was the most important component of mineral N under all heaps due to limited oxygen diffusion to the soil. N leaching and end product quality were affected by a combination of treatment option (i.e. storage and cover) and initial manure characteristics. When CFM was characterized by a low volumetric moisture content and high C/N ratio, so in case of straw-rich CFM or CFM with added bulking agents, composting led to the least N leaching and most stable end product. When CFM was characterized by a high volumetric moisture content and low C/N ratio, stockpiling and covering (plastic or geotextile) resulted in lower N leaching to the soil. Stockpiling and covering the CFM with a geotextile resulted in a more stable end product than did covering with a plastic.

5.1. Introduction

On the one hand, cattle farmyard manure (CFM) might be an important soil improver that adds large amounts of organic matter (OM) to the soil (Vanden Nest et al., 2014), which is especially valuable in regions with declining soil organic carbon contents. Moreover, CFM is a valuable resource as it can be used to recycle nutrients to the soil, thus reducing the use of mineral fertilizers. On the other hand, storage and application of CFM can cause considerable environmental problems through nutrient losses to soil, water and air, if not properly handled (Seymour and Bourdon, 2003; Rotz, 2004). Despite the increasing interest in limiting the emissions from farming operations, the calculated total nitrogen (N) loss in 2008 from agriculture in the EU-27 was still 13 Mt N, with 53% as N_2, 22% as nitrate (NO_3^-), 21% as ammonia (NH_3), 3% as nitrous oxide (N_2O) and 1% as mono-nitrogen oxide (NO_x; Velthof et al., 2014). Currently, to prevent nutrient losses, Flemish farmers are not allowed to store solid manure on the field between 15 November and 15 January, and no longer than 2 months during the rest of the year (fifth Manure Action Plan). The regulation restricts manure storage on a concrete floor, including the capture of rainwater and leachate. However, this means of storage is an investment for the farmer and often leads to a suboptimal manure quality, since the manure is captured between concrete walls and the lower part of the heap will become too wet, causing rotting and nutrient leaching, while the upper part will become too dry, causing internal heating and volatile losses (Kolenbrander and De La Lande Cremer, 1967). The challenge is to store and apply CFM in a simple and cost-effective way, as individual farmers generally cannot afford time-consuming, labor- or machine-intensive operations, thereby reducing the environmental risks and, at the same time, improving or at least maintaining the quality as soil improver. Alternative methods for uncovered stockpiling are *covering* the CFM during storage, either with an airtight plastic cover or with a geotextile that prevents infiltration of rainwater, while allowing gas exchanges. *Composting* the CFM can mitigate nutrient losses in the field during application and further decomposition, as they are stabilized in an organic form (Francou et al., 2005; Annabi et al., 2007; Bernal et al., 2009). However, a potential disadvantage is the high risk for volatile N losses during composting (Eghball et al., 1997), which are not only posing an environmental problem but also reducing the amount of nutrients available for plant growth. *Co-composting* the CFM can reduce nutrient losses during composting by increasing the water-holding capacity and the C/N ratio of the pile (Krogmann and Körner, 2000) by adding

bulking agents such as straw (Ulén, 1993) and wood chips (Lafrance et al., 1996). However, those bulking agents are expensive or not always available at the farm (Viaene et al., 2016b).

To the best of our knowledge, this is the first work that compares N leaching and product quality for different treatment options with different types of CFM at a field scale. To do so, three consecutive field-scale experiments were conducted with CFM stored and treated on the headland. In the first experiment, we compared the decomposition process, N leaching and product quality for (1) uncovered stockpiling versus (2) extensive composting of CFM, in combination with covering with a geotextile. On the basis of the findings of the first study, a second experiment was executed to even better understand the processes at play and the detailed differences among treatments. During this second experiment, the following four treatments were compared: (1) uncovered stockpiling, (2) covering with plastic cover, (3) covering with a geotextile and (4) composting combined with covering with a geotextile. The set-up was repeated on three different locations with three different types of CFM to investigate the effect of CFM on N leaching. In a third experiment, the effect of adding bulking agents (hay) from nature-oriented grassland management, which are abundantly available in the region, on N leaching and product quality in composting CFM was tested. The following three treatments were compared: (1) composting CFM; (2) co-composting CFM with hay of grass; and (3) co-composting CFM with pre-composted hay of grass (feedstock mixture of old hay of grass, more coarse hay, wood chips and fresh grass clippings).

5.2. Materials and Methods

The weather data (temperature and precipitation) were used from the KMI meteorological station in Melle, Belgium.

5.2.1. Treatments

The specifications of the three experiments are summarized in Table 5.1.

Table 5.1 Specifications of the three field experiments.

Experiment	Duration	Location	Code	Treatments	Cover	Turning events
1	17/04/2014 - 17/06/2014	Merelbeke	S1	Uncovered stockpiling	No cover	No turning
	60 days		C1	Composting	TopTex	Day 1, day 29
2	15-18/12/2014 - 11-19/02/2015	Zwevezele	S2	Uncovered stockpiling	No cover	No turning
	54-64 days	Zoersel	ST2	Covered storage	TopTex	No turning
		Wachtebeke	SP2	Covered storage	Plastic + TopTex	No turning
			C2	Composting	TopTex	Day 1, day 28 or 29 or 31
3	14/04/2015 - 10/06/2015	Molenstede	C3	Composting	TopTex	Day 3, 27 and 48
	56 days		CG3	Co-composting with 8-months old grass clippings	TopTex	Day 3, 27 and 48
			CC3	Co-composting with pre-composted old grass clippings	TopTex	Day 3, 27 and 48

5.2.1.1. Experiment 1

The first field experiment started on 17 April and lasted until 17 June 2014 (60 days), and took place at the Institute of Agricultural and Fisheries Research (ILVO) in Merelbeke, Belgium. A straw-rich CFM was removed from a deep-litter stable with dairy cattle at ILVO, 1 week before set-up. The manure was homogenized using a manure spreader before the start of the experiment. Two CFM treatments with approximately the same volume (24 m^3) were compared, namely, uncovered stockpiling (*S1*) and extensive composting in combination with covering with a geotextile (TopTex®, TenCate, Almelo, The Netherlands) (*C1*) on the headland (WRB classification: Bathygleyic Cambisol; Dondeyne et al., 2014). With S1, CFM was stored on a surface of 24 m^2 (8 m length and 3 m width). With C1, the CFM was put on a windrow with the same dimensions and turned with a compost turner (Type TG 301, Gujer Innotec AG, Mesikon, Switzerland) to homogenize and aerate the CFM. After turning, the windrow was covered with a TopTex cover. After 1 month, the compost pile was turned again to bring the outer material from the surface to the core of the pile to ensure degradation of the whole pile. We chose to have an extensive composting system (no addition of water and only turning twice) to decrease labor costs for the farmer and the risk for gaseous losses (Sommer and Dahl, 1999).

5.2.1.2. Experiment 2

The second field experiment was set up between 15-18 December 2014 and lasted until 11-19 February 2015 (between 54-64 days), at three different locations in Flanders: Zwevezele (WRB classification: Terric Anthrosols), Zoersel (WRB classification: Eutric Fluvic Endogleyic Arenosol) and Wachtebeke (WRB classification: Terric Cambisols (Arenic)). Soil classifications were according to Dondeyne et al. (2014). The used beef CFM at the different locations was removed from the litter loose houses with sloping floor less than one month before setup. Four CFM treatments starting from the same volume (24 m^3 each) were compared at each location: uncovered stockpiling (*S2*), storage with a TopTex cover (*ST2*), storage with a plastic cover and TopTex cover (to prevent bird picking) (*SP2*), and composting in combination with covering with TopTex cover (*C2*). The TopTex cover allows gas exchanges, while the plastic cover ensures a semi-anaerobic environment. For S2, ST2 and SP2, the CFM was slightly pressed with a telescopic handler after storage, resulting in a shape similar to C2 allowing rainwater to run from the piles. For C2, the CFM was put on a pile using a manure spreader and turned after one month using a compost turner (Type 3300 or 4300 SP, Ménart, Dour, Belgium).

5.2.1.3. Experiment 3

The third field experiment was conducted from 14 April until 10 June 2015 (56 days) in Molenstede, Belgium (WRB classification: Endogleyic Brunic Umbrisol (Arenic), according to Dondeyne et al. (2014)). The used beef CFM was removed from a deep-litter stable less than 1 month before the start of the experiment. The following three CFM treatments were compared: composting CFM (*C3*), co-composting CFM with approximately 8-month-old hay of grass (stored in bales) from a neighboring nature reserve (*CG3*) and co-composting CFM with precomposted hay of grass (*CC3*). The same hay as used in CG3 was pre-composted in the nature reserve for 1 month before the experiment to reduce volume and, thus, transport costs. To obtain an optimal starting mixture for this pre-composting, other byproducts from nature reserve management were added to the hay (40% v/v), i.e. wood chips (20% v/v), more coarse hay (from ferns; 20% v/v) and fresh grass clippings (from soft rush; 20% v/v). The CFM was added at two times, i.e. at the start of the experiment and after 21 days, which is in line with routine farm practices. The final compositions of C3, CG3 and CC3 were (1) 100% v/v CFM (45m^3), (2) 44% v/v CFM (17.5 m^3) and 56% v/v hay (22.5 m^3), and (3) 44% v/v CFM (17.5 m^3) and 56% v/v pre-composted hay (22.5 m^3) respectively. The piles were turned at the start of the experiment, after 1 month and 48 days using a compost turner (Type 4300 SP, Ménart).

5.2.2. N leaching to the soil

5.2.2.1. Experiment 1

For S1 and C1 soil analyses (0-30, 30-60 and 60-90 cm layers) were conducted before the experiment, after 32 days and after 61 days. One mixed sample (mixture of 16 subsamples) over the whole length of the piles was taken on four positions: under the piles (by removing part of the manure after 32 days, and after removing the whole pile at day 60), at the border of the piles (under the TopTex cover for C1), 30 cm from the piles (just outside the TopTex cover for C1) and 5 m from the piles. The position of the soil samples is illustrated in Appendix Figure A2. The soil samples were analyzed for mineral N (NO_3^--N and NH_4^+-N) content (ISO 14256-2, Skalar SAN++ Flow Analyzer, Skalar Analytical B.V., Breda, The Netherlands).

5.2.2.2. Experiment 2

At the start of the second experiment, four (one per pile) mixed samples (from 16 subsamples) at each location were taken from four soil layers (0-10, 10-30, 30-60 and 60-90 cm). Furthermore, soil (0-30 cm) was taken from the three locations to use in an incubation experiment, and analyzed for pH-KCl (ISO 10390), total organic carbon (TOC) (ISO 10694) and total N content (Dumas ISO 13878, flash 4000 total N analyser, Thermo Fisher Scientific, MA, USA). At the end of the experiment, three mixed samples (from eight subsamples) were taken of the respective soil layers (0-10, 10-30, 30-60 and 60-90 cm) under the piles after removing the CFM, one mixed sample (from eight subsamples) was taken at 30 cm from the piles (outside the cover for ST2, SP2 and C2). Furthermore, three mixed samples (from eight subsamples) per layer were taken around the four piles, at more than 1 m distance as a reference (Appendix Figure A2). All soil samples were analyzed for mineral N content (ISO 14256-2, Skalar SAN++ Flow Analyzer).

On the basis of the elevated soil N concentrations we observed in Experiment 1, two plastic containers (3.6 L), covered with a perforated lid and filled with 100 g clinoptilolite, were buried under each pile, to capture the potential leachate losses under the piles. Clinoptilolite, a zeolite with a high cation exchange capacity (Inglezakis et al., 2004), was used to adsorb NH_4^+-N in the liquid, thereby reducing potential NH_3 losses. The position of the containers is illustrated in Appendix Figure A3. At the end of the experiment, the leachate was analyzed for volume, organic and NH_4^+-N content (Kjeldahl-N derived from ISO 5983-2, Gerhardt Vapodest VAP50 UT carousel, C. Gerhardt GmbH & Co. KG, Königswinter, Germany) and NO_3^--N content (Skalar SAN++ Flow Analyzer). Total N content adsorbed by the clinoptilolite was determined according to Dumas (EN 13654-2). Subsequently, the total amount of N collected in the containers could be calculated as the sum of the amounts of organic N, NH_4^+-N and NO_3^--N measured in the liquid and the amount of N adsorbed to the clinoptilolite.

Furthermore, soil temperatures at a depth of 10 cm in the middle of the piles (Appendix Figure A3) were monitored for ST2 and C2, and further away from the piles (reference temperature) with an EBI 20-T1 temperature logger (Xylem Analytics, ebro, Ingolstadt, Germany). Simultaneously with the field experiments, an incubation trial in the lab was set up to gain insight in the soil N mineralization process under the piles under different soil temperature conditions. The soil was incubated at 70% RH and at two different temperatures (three replications per location, temperature and time): a temperature in accordance with the

temperature under a stored CFM pile (higher temperature) and a normal soil temperature (reference temperature). Soil temperature was measured at the start of the experiment and the incubation temperatures were adjusted according to the weekly measured soil temperature under ST2 in Wachtebeke. Soil was placed in PVC tubes (Ø 4.63 cm, filling height 12 cm and bulk density 1.4 g cm^{-3}). Before filling the tubes, demineralized water was added to obtain a gravimetric moisture content of 16.8% (w/w) equivalent to 50% water-filled pore space. After thorough mixing, the tubes were filled and covered with a single layer of gas-permeable Parafilm® M Barrier Film (Pechiney Plastic Packaging, Chicago, IL, USA) to minimize water loss. Entire tubes were destructively sampled and soil mineral N content of the samples was measured at the start of the incubation experiment and after 4 and 8 weeks, according to ISO 14256-2 (Skalar SAN++ Flow Analyzer).

5.2.2.3. Experiment 3

At the start of the third experiment, three (one per pile) mixed soil samples (from eight subsamples) were taken (0-10, 10-30, 30-60 and 60-90 cm soil layers). At the end of the experiment, three mixed samples (from eight subsamples) over the whole length of the piles were taken under the piles after removing the CFM, at 30 cm from the piles (just outside the TopTex cover) and at more than 10 m distance from the piles as a reference. All soil samples were analyzed for mineral N content (ISO 14256-2, Skalar SAN++ Flow Analyzer).

Furthermore, similar to Experiment 2, two containers (3.6 L), covered with a perforated lid were buried under each pile, to capture the potential liquid losses under the piles. However, based on the first experience with these containers in Experiment 2, we concluded that it was not necessary to add clinoptilolite to the containers to adsorb NH_4^+-N in the liquid. At the end of the experiment, the liquid was analyzed for volume, organic and NH_4^+-N content (Kjeldahl-N derived from ISO 5983-2) and NO_3^--N content (Skalar SAN++ Flow Analyzer). Soil temperatures at a depth of 10 cm in the middle of the piles and at 20 m from the piles (reference temperature) were monitored with an EBI 20-T1 temperature logger (Ebro).

5.2.3. Agricultural value of the end products

5.2.3.1. Experiment 1

The temperature (Digital Thermometer GTH1150, GHM Messtechnik GmbH Standort Greisinger, Regenstauf, Germany) and CO_2 levels (CO2-Indicator TESTORYT, BRIGON Messtechnik GmbH, Rodgau, Germany) were monitored manually three to four times per

week on four different points per pile. To analyze the product quality, samples were taken at the start of the experiment, after 32 days and at the end of the experiment (after 60 days). Per treatment four mixed samples were taken: CFM over the whole length of the pile was collected in a container of ca. 0.8 m³, after mixing, one sample was taken and this was repeated four times. Samples were analyzed for physico-chemical compost quality parameters: fresh bulk density and dry matter (DM) content (EN 13040), OM content (EN 13039), pH-H_2O (EN 13037), electrical conductivity (EC) (EN 13038), mineral N content (NH_4^+-N and NO_3^--N, ISO 14256-2, Skalar SAN++ Flow Analyzer) and total N content (Dumas EN 13654-2, flash 4000 total N analyzer, Thermo Scientific). Total Ca, K, Mg and Na concentrations were measured by a charge-coupled device (CCD) simultaneous Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES; VISTA-PRO, Varian, Palo Alto, CA, USA) and total P was measured with a Varian CARY 50 Spectrophotometer, after ashing and digestion with 7N HNO_3. Product stability, preventing negative impacts on plant growth when added to soil, was evaluated by oxygen uptake rate (OUR), biodegradation potential, NO_3^--N/NH_4^+-N ratio and a germination test. A lower OUR and biodegradation potential and a NO_3^--N/NH_4^+-N ratio > 1 indicate more stable products (Zucconi and de Bertoldi, 1987). The OUR was calculated from the oxygen consumption measured by shaking 20 g fresh product in 200 mL buffered nutrient solution during five days at 20°C in a closed OxiTop® respirometer (Global Water Instrumentation, TX, USA), in which the CO_2 was trapped by Sodalime (Merck Millipore, Darmstadt, Germany); the pressure drop in this condition is only due to the oxygen consumption because of the microbial respiration (Grigatti et al., 2007). OUR was expressed as mmol kg^{-1} OM h^{-1}. Furthermore, the cell wall components were measured according to the method of Van Soest et al. (1991). The biodegradation potential of an organic material can then be estimated by the (hemicellulose+cellulose)/lignin ratio with hemicellulose = neutral detergent fiber (NDF) – acid detergent fiber (ADF), lignin = acid detergent lignin (ADL) and cellulose = ADF-ADL (Van Soest et al., 1991). For the germination test, 500 mL product was mixed with 2000 mL white peat and spread in a layer of 2-3 cm in a container, and kept for three weeks at 21°C and 100% RH under conditions of sufficient natural light. The number of emerging weeds was counted after three weeks.

5.2.3.2. Experiments 2 and 3

Pile temperatures (Digital Thermometer GTH1150) were monitored one to two times per week on four different points per pile. To analyze the product quality, samples were taken

at the start (one and four mixed samples per pile for Experiment 2 and 3, respectively) and at the end of the experiment (three and four mixed samples per pile for Experiment 2 and 3, respectively). Samples were analyzed for physico-chemical compost quality parameters as described in Experiment 1.

5.2.4. Statistical analysis

For Experiment 1, an independent-samples *t*-test at $p = 0.05$ was used to assess the significant differences between the values of each parameter of S1 and C1. For Experiment 2, the two-way analysis of variance (ANOVA) with factors 'treatment' and 'location' at $p = 0.05$ was used to assess the significant differences for the manure treatments. In the case of a significant interaction term ($p < 0.05$), a one-way ANOVA was conducted for each location with 'treatment' as factor. In case 'treatment' or 'location' was significant, a post-hoc Scheffé test was conducted to compare the effect of the individual levels of the factor treatment or location. For certain parameters (e.g., OM content, C/P ratio and OUR), the start versus end value was statistically tested using a repeated-measures design. For Experiment 3, a one-way ANOVA with factor 'treatment' at $p = 0.05$ was used to assess the significant differences for the manure treatments. To test the differences in soil mineral N concentrations, a one-way ANOVA with post-hoc Scheffé test at $p = 0.05$ was used. So as to assess the product quality, two multiple linear regressions (stepwise method) at $p = 0.05$ were performed with the average data of all three experiments, with the pile temperature as dependent variable and the initial or end product parameters respectively as independent variables. Data analysis was carried out using IBM SPSS Statistics for Windows, Version 21.0 (IBM Corp., Armonk, NY, USA).

5.3. Results

5.3.1. Experiment 1: composting versus uncovered stockpiling

In Experiment 1 we tested if extensive composting CFM (+ TopTex cover) resulted in less soil mineral N under the piles and a more stable end product compared to uncovered stockpiling.

5.3.1.1. Process monitoring

The mean ambient temperature and total precipitation during the field experiment were 13.9°C and 145 mm, respectively. Both piles increased in temperature due to microbial activity directly after the start of the experiment and reached a maximum temperature of 65-70°C (thermophilic range) during the first 10 days (Appendix Figure A4). The temperature of S1 was on average 38% higher than C1 during the full duration of the experiment. After 60 days, S1 still had a temperature of 43°C, while C1 reached a temperature around 30°C. The CO_2 concentrations were always below 3%, indicating sufficient oxygen supply during the experiment (data not shown).

5.3.1.2. N leaching

Figure 5.1 shows the soil NO_3^--N (Figure 5.1B), NH_4^+-N (Figure 5.1A) and total mineral N (NO_3^--N + NH_4^+-N) (Figure 5.1C) concentrations in the 0-30, 30-60 and 60-90 cm layers for S1 and C1 during the experiment. The differences in mineral N concentrations among the treatments could not be tested statistically since there were no replicates. At the start of the experiment, there was no indication that there were differences in soil mineral N concentrations among the different positions. Generally, during the experiment, the NH_4^+-N concentrations under the piles were higher in the top soil layer (0-30 cm) than those in the 30-60 and 60-90 cm layers (Figure 5.1A). Furthermore, the total mineral N concentration (Figure 5.1C) at the end of the experiment in the 0-30 cm layer was seven times higher under S1 compared to C1, related to the higher NH_4^+-N concentration (Figure 5.1A), being 483 and 68 kg N ha^{-1} for S1 and C1, respectively. This trend was also seen in the 0-30 cm layer at the border of the piles, with the NH_4^+-N concentrations being 43 and 20 kg NH_4^+-N ha^{-1}, respectively for S1 and C1. Also in the 30-60 cm layer under the piles a higher NH_4^+-N concentration was measured under S1 compared to C1 (23 and 8 kg N ha^{-1}, respectively. The NO_3^--N concentrations in the 0-30 and 30-60 cm layers were < 3 kg N ha^{-1} under and at the border of the piles (Figure 5.1B).

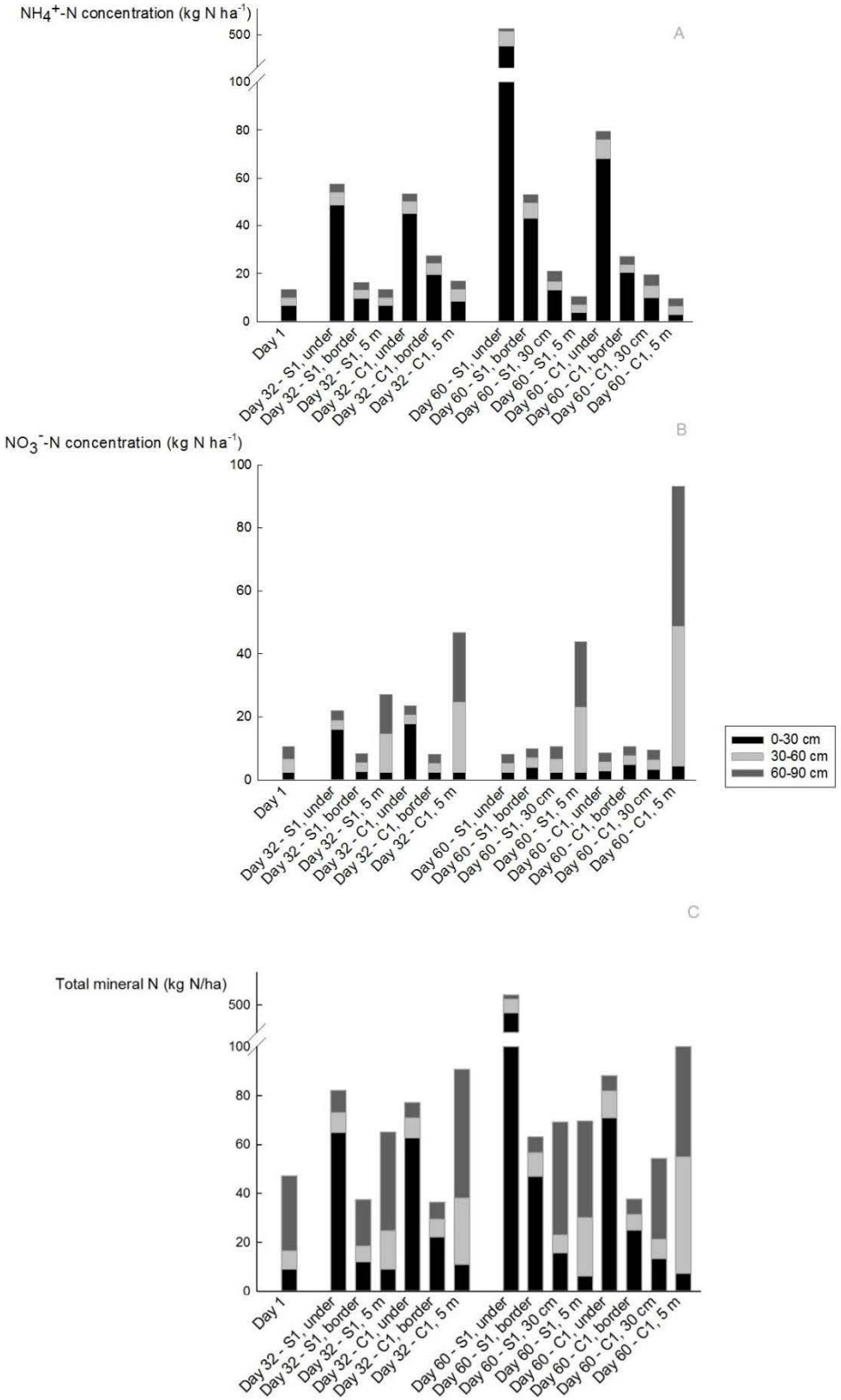

Figure 5.1 Soil NH_4^+-N (A), NO_3^--N (B) and total mineral N (NO_3^--N + NH_4^+-N) (C) concentrations in the 0-30, 30-60 and 60-90 cm layers at the start of Experiment 1, and after 32 and 60 days, for the stored (S1) and composted (C1) cattle farmyard (n = 1).

5.3.1.3. Agricultural value of the products

Table 5.2 summarizes the results of the analyzed parameters of S1 and C1 at the start of the experiment, after 32 and 60 days.

Table 5.2 Product quality for stored (S1) and composted (C1) cattle farmyard manure at the start of Experiment 1, at day 32 and day 60 (mean ± standard deviation, n = 4). Numbers in bold are significantly different (t-test, $p < 0.05$). DM = dry matter, FM = fresh matter and OM = organic matter, nda = no data available.

	Day 1	Day 32			Day 60		
	S1 and C1	S1	C1	p -value	S1	C1	p -value
Fresh bulk density (kg m^{-3})	169 ± 5	**91 ± 15**	**139 ± 29**	0.027	**183 ± 32**	**125 ± 10**	0.016
pH-H_2O (-)	8.2 ± 0.1	9.6 ± 0.1	9.8 ± 0.0	0.179	9.7 ± 0.1	9.7 ± 0.1	0.756
Electrical conductivity (µS cm^{-1})	2635 ± 89	**637 ± 61**	**983 ± 62**	0.000	1230 ± 89	1158 ± 37	0.189
Organic matter content (% of DM^{-1})	83.4 ± 0.4	78.6 ± 3.8	76.0 ± 1.2	0.231	73.4 ± 2.2	72.0 ± 1.0	0.269
Dry matter content (% of FM^{-1})	32.2 ± 0.2	40.8 ± 5.3	46.2 ± 1.1	0.099	**42.0 ± 2.8**	**56.6 ± 0.9**	0.000
Volumetric moisture content (kg m^{-3} FM)	114 ± 4	54 ± 14	75 ± 16	0.137	**115 ± 28**	**54 ± 5**	0.007
NO_3^--N (mg kg^{-1} DM)	4 ± 0	20 ± 9	19 ± 3	0.923	339 ± 189	303 ± 78	0.746
NH_4^+-N (mg kg^{-1} DM)	518 ± 40	550 ± 336	128 ± 20	0.086	203 ± 66	123 ± 18	0.089
NO_3^--N / NH_4^+-N (-)	0.007 ± 0.001	**0.039 ± 0.035**	**0.153 ± 0.046**	0.005	1.815 ± 1.146	2.542 ± 0.843	0.346
N (g kg^{-1} DM)	18.4 ± 0.6	23.8 ± 1.6	23.6 ± 1.7	0.853	**29.8 ± 2.3**	**26.3 ± 1.3**	0.039
P (g kg^{-1} DM)	3.3 ± 0.2	4.1 ± 0.3	4.3 ± 0.4	0.524	5.1 ± 0.5	4.7 ± 0.1	0.256
C/N (-)	25.3 ± 0.9	18 ± 2	18 ± 1	0.697	14.4 ± 0.9	15.2 ± 0.8	0.237
C/P (-)	139 ± 8	107 ± 11	99 ± 11	0.399	81 ± 9	85 ± 2	0.531
K (g kg^{-1} DM)	27.3 ± 0.2	39.5 ± 5.7	40.5 ± 1.5	0.748	**48.1 ± 2.9**	**42.6 ± 1.3**	0.014
Ca (g kg^{-1} DM)	7.7 ± 0.4	9.8 ± 0.4	10.6 ± 0.8	0.130	11.7 ± 0.8	10.9 ± 0.2	0.147
Oxygen Uptake Rate (mmol kg^{-1} OM h^{-1})	29.1 ± 17.4	16.9 ± 3.5	14.3 ± 8.4	0.725	11.3 ± 4.2	8.3 ± 1.9	0.444
Hemicellulose (% of OM)	16.0 ± 0.8	11.1 ± 7.0	9.3 ± 1.7	0.071	**7.8 ± 5.1**	**8.8 ± 1.2**	0.026
Cellulose (% of OM)	28.9 ± 0.8	21.7 ± 4.7	17.9 ± 1.2	0.134	6.8 ± 3.6	9.6 ± 1.1	0.191
Lignin (% of OM)	9.3 ± 0.5	10.6 ± 0.4	10.1 ± 0.4	0.122	**17.6 ± 1.3**	**15.4 ± 0.7**	0.027
Biodegradation potential (-)	4.8 ± 0.3	3.1 ± 0.6	2.7 ± 0.2	0.094	0.8 ± 0.3	1.2 ± 0.1	0.064
Germinal weed seeds (number L^{-1})	nda	nda	nda		0 ± 0	0 ± 0	

At the start of the experiment the OM content of the CFM was 83.4% of DM, but no significant differences were found between the end products of S1 and C1. The CFM at the start was characterized by a low fresh bulk density (169 kg m^{-3}) and high DM content (32.2% of fresh matter, FM), due to the high straw dose and homogenization by the manure spreader. The end product of C1 was significantly drier than that of S1, explained by turning the compost (higher water losses) and covering the pile (preventing rainwater to enter). When converting the gravimetric DM content to the volumetric moisture content by using the bulk density, composting resulted in a significantly lower volumetric moisture content (54 kg m^{-3} FM), meaning less moisture per volume of manure, than with uncovered stockpiling (115 kg m^{-3} FM). The C/N ratio of the CFM was 25 at the start of the experiment and decreased the most during the first 32 days due to microbial activity. Total K content was higher for S1 compared to C1. After 32 days, the total N content and NO_3^--N/NH_4^+-N ratios increased and the NO_3^--N/NH_4^+-N ratio was significantly higher for C1 than for S1. It can be concluded that the degradation process in S1 was lower than in C1 after 32 days. After 60 days, there were no differences anymore in mineral N concentrations between S1 and C1 (339 and 303

mg NO_3^--N kg^{-1} DM; 203 and 123 mg NH_4^+-N kg^{-1} DM, respectively for S1 and C1). In contrast, the total N content was significantly lower for C1 (26.3 g kg^{-1} DM) compared to S1 (29.8 g kg^{-1} DM). The NO_3^--N/NH_4^+-N ratios were higher than 1 for S1 and C1, indicating stabilization of the manure in both treatments. The initial CFM had an OUR of 29.1 ± 17.4 mmol kg^{-1} OM h^{-1}, which decreased over time for both treatments, indicating more stabilized products after the process. Generally, the percentage hemicellulose and cellulose (on an OM base) decreased over time, while the percentage lignin increased, leading to a decreased biodegradation potential in all treatments. After 60 days, the percentage hemicellulose was significantly higher for C1 than S1, but there was no difference in biodegradation potential. No emerging weeds were detected.

From Experiment 1 we can conclude that uncovered stockpiling of CFM resulted in higher pile temperatures and higher soil mineral N concentrations (especially NH_4^+-N) under the piles in the 0-30 cm soil layer than with composting. However, on the basis of this experiment, we cannot deduct the relative importance of either direct leaching or increased soil mineralization due to higher soil temperatures to explain the higher soil mineral N content. End product stability was similar for both treatments on the basis of evaluation of C/N ratio, NO_3^--N/NH_4^+-N ratio, OUR and biodegradation potential. However, after composting, the end product was drier, contained less total N, more total K and relatively more hemicellulose and less lignin than after uncovered stockpiling.

5.3.2. Experiment 2: effect of CFM type and storage method

In Experiment 2 there were three main research questions: (1) Can the results of Experiment 1 be extrapolated to different types of CFM, (2) What are the processes explaining the higher soil N concentrations under the piles, and (3) Can a simple covering of the CFM (with TopTex or plastic) be a suitable, less labor-intensive practice than composting, in terms of N leaching and product quality?

5.3.2.1. Process monitoring

The mean ambient temperature and total precipitation during the field experiment were 4°C (of which six days were < 0°C) and 190 mm, respectively. Appendix Figure A5 shows the temperature profiles of the different treatments per location. In contrast to the composting treatment in Experiment 1, C2 was the treatment with the highest temperatures at all locations, while SP2 was the treatment with the lowest temperatures. ST2 resulted in higher temperatures than did SP2 and S2. Generally, the pile temperatures were the highest in

Wachtebeke and the lowest in Zwevezele. The piles in Wachtebeke showed an increase in temperature directly after the start of the experiment and reached a maximum temperature during the first 15 days. Moreover, only C2 in Wachtebeke showed temperatures above 55°C, indicating a good composting process. The lower temperatures of C2 (maximum 44°C in Zoersel, and 29°C in Zwevezele) were pointing to a suboptimal composting process.

5.3.2.2. N leaching

Soil characterization and temperatures

The soil characteristics are given in Appendix Table A7. The soil temperatures under the piles (Appendix Figure A6) followed the same trend as the pile temperatures; the highest and lowest temperatures were observed in Wachtebeke and Zwevezele, respectively. The soil temperature under C2 was higher than under ST2 in all three locations, and reached 37°C in Wachtebeke.

Incubation experiment

The NH_4^+-N concentrations during the incubation experiment were very low (< 2.5 kg NH_4^+-N ha^{-1}), indicating aerobic circumstances. The NO_3^--N concentrations (Figure 5.2) were higher at a higher soil temperature than the reference soil temperature. In the case of Wachtebeke an extra 9.0 g NO_3^--N m^{-2} was mineralized after 56 days of incubation, assuming that the temperature raise was until 20 cm deep (sensor was placed at 10 cm soil depth). In the case of Zoersel and Zwevezele, the difference was 6.4 g NO_3^--N m^{-2}.

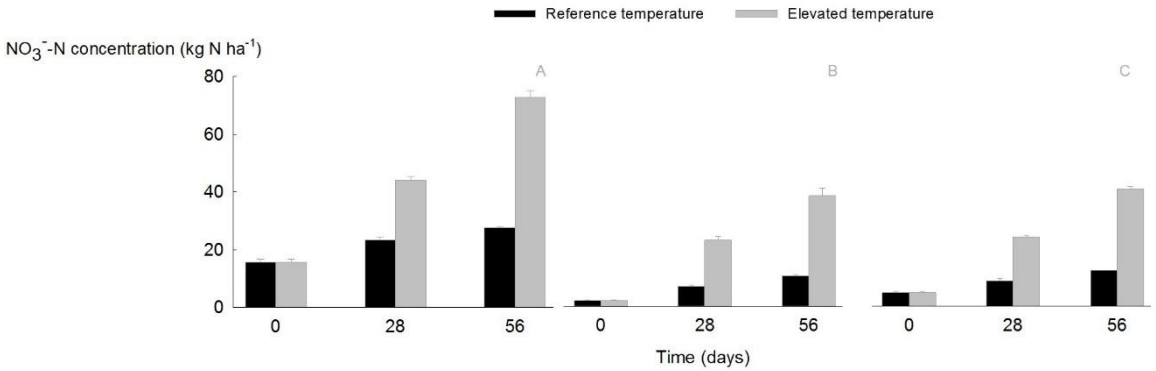

Figure 5.2 NO_3^--N concentration in soil incubated at a reference temperature (similar to the temperature of soil not covered by a manure pile) and an elevated temperature (similar to the temperature under a manure pile covered with TopTex) in Experiment 2 in Wachtebeke (A), Zoersel (B) and Zwevezele (C).

N leachate losses under the piles

Figure 5.3 shows the total amount of N in the leachates collected in the containers under the piles, ranging from 0.1 to 15.0 g per container. The leachate volume per container ranged from 0 (ST2 in Zwevezele) to a completely filled container (3475 mL for SP2 in Zoersel, full container so potentially even more), but variability was high; often there was a large variation in leachate volume between the two replicates of the treatment. In most containers, the leachate N was dominated by organic N (on average 65 ± 21% of the Kjeldahl-N content).

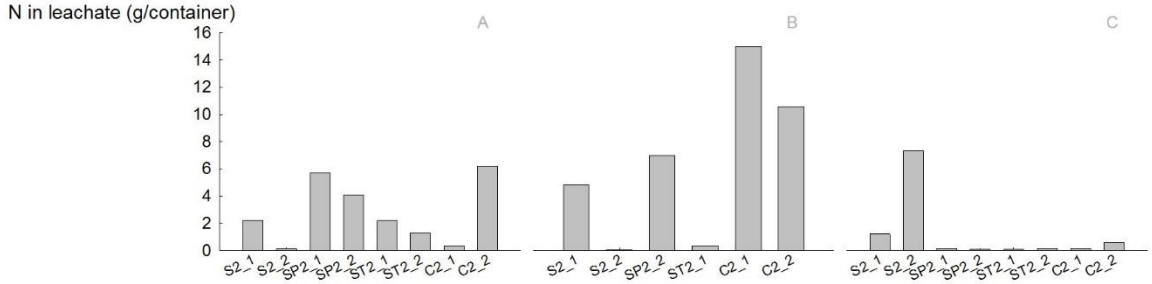

Figure 5.3 Amount of N in leachate losses at the end of Experiment 2 (total of adsorbed N to the clinoptilolite and NH_4^+-N, NO_3^--N and organic N in the leachate) collected in the containers under the manure piles in Wachtebeke (A), Zoersel (B) and Zwevezele (C). Per treatment, two containers were sampled for stored (S2_1 and S2_2), covered with plastic (SP2_1 and SP2_2), covered with TopTex (ST2_1 and ST2_2) and composted (C2_1 and C2_2) cattle farmyard manure. Two data points are missing in Zoersel (S2_1 and ST2_2) since two containers were broken.

Soil mineral N

The total mineral N (Figure 5.4c) and NH_4^+-N (Figure 5.4a) concentrations under the piles were much higher in the 0-10 cm layer than in the deeper soil layers; moreover, they were much higher than in the reference 0-10 cm soil layer without manure piles. The total mineral N (Figure 5.4c, A) and NH_4^+-N (Figure 5.4a, A) concentration in the 0-10 cm layer in Wachtebeke was higher under C2 compared to SP2 and ST2, but did not differ from that in S2. In Zoersel, the mineral N (Figure 5.4c, B) and NH_4^+-N (Figure 5.4a, B) concentration was higher under C2 than in all other treatments, while in Zwevezele it was higher under C2 than in the ST2 only (Figure 5.4a and c, C). There was no difference in NH_4^+-N and total mineral N concentration among the non-composted treatments (S2, SP2 and ST2) in the 0-10 cm and 10-30 cm layers on all three locations. The NH_4^+-N concentration in Wachtebeke in the 0-30 cm layer under S2 was approximately 300 kg ha^{-1}, compared with approximately 200 kg ha^{-1} in Zoersel and Zwevezele. Under C2 in Wachtebeke and Zoersel, it amounted to

500 kg N ha^{-1}. The apparent amount of leached N in the 0-90 cm layer under the piles can be calculated as follows: $N_{leached\ under} = (N_{under} - N_{ref} - N_{inc}) * pile\ area$. With $N_{leached\ under}$ = apparent amount of leached NH_4^+-N under the pile to the 0-90 cm layer [kg N/pile]; N_{under} = soil NH_4^+-N concentration in the 0-90 cm layer under the pile [kg N m^{-2}]; N_{ref} = soil NH_4^+-N concentration in the 0-90 cm layer of the reference soil [kg N m^{-2}], N_{inc} = mineralized amount of NH_4^+-N due to elevated temperatures in the 0-20 cm layer (incubation experiment) [kg N m^{-2}] and pile area = soil surface covered with the manure pile [m^2/pile]. When using the minimal (S2 in Zoersel) and maximal (C2 in Wachtebeke) soil N concentration, 0.28 - 1.13 kg NH_4^+-N could leach from the pile to the 0-90 cm layer. This equals 0.3 - 2.2% of the initial N content of the manure at the start of the experiment that was lost through leaching.

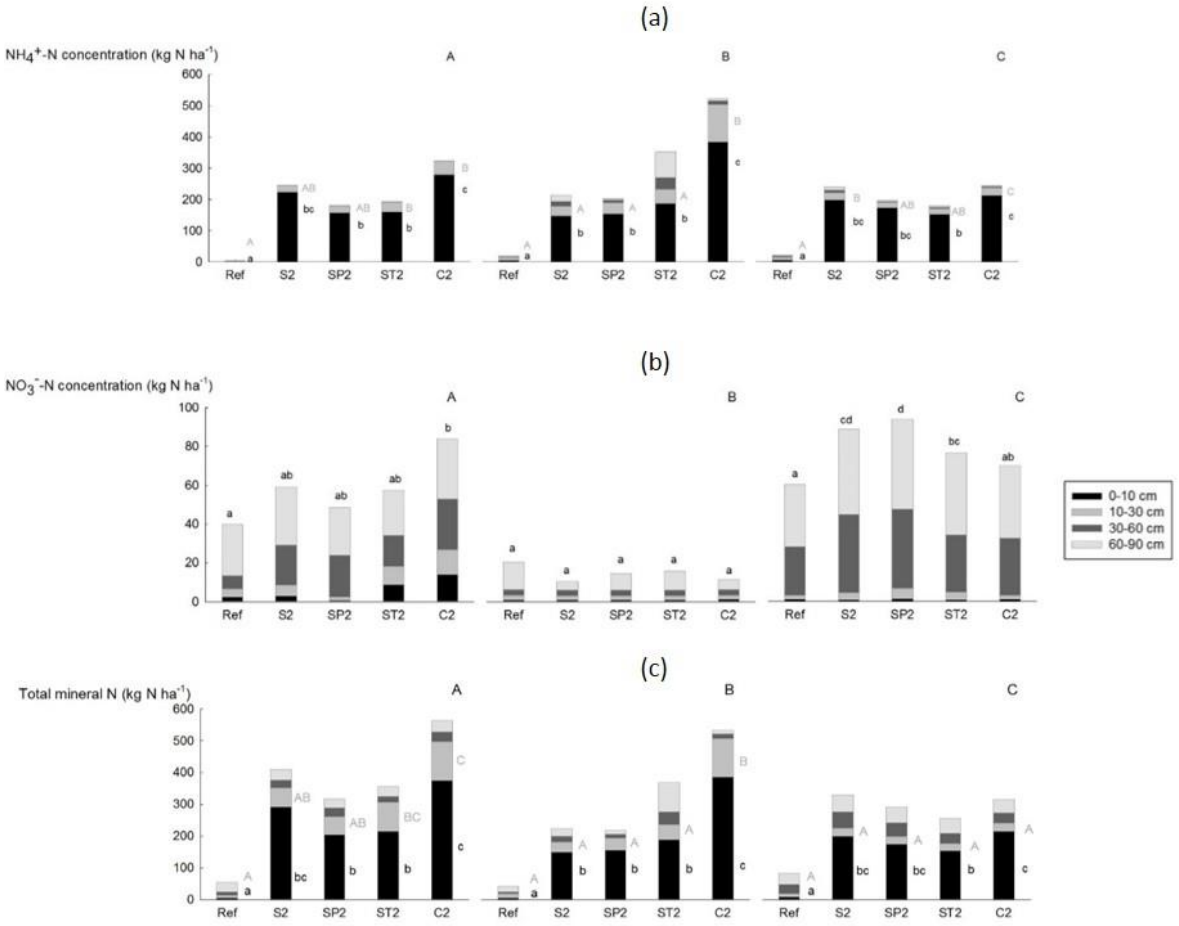

Figure 5.4 (a) NH_4^+-N, (b) NO_3^--N and (c) total mineral N (NH_4^+-N + NO_3^--N) concentration (n = 3) in the 0-10, 10-30, 30-60 and 60-90 cm soil layer under the piles at the end of Experiment 2 for a reference situation (soil not covered by a manure pile) and under stored (S2), covered with plastic (SP2), covered with TopTex (ST2) and composted (C2) cattle farmyard manure in Wachtebeke (A), Zoersel (B) and Zwevezele (C). 0-10 and 10-30 cm soil layers with the same letter are not significantly different (Scheffé test per layer, $p < 0.05$).

5.3.2.3. Agricultural value of the products

Table 5.3 summarizes the quality of the end products, standard deviations are given in Appendix Table A8.

The fresh bulk density of the CFM at the start of the experiment was lower in Wachtebeke (347 kg m^{-3}) than in Zoersel (599 kg m^{-3}) and Zwevezele (656 kg m^{-3}). The CFM at the start of the experiment was drier (higher DM content) in Wachtebeke (22.8% of FM) than in Zwevezele (19.9% of FM). The three types of CFM differed in volumetric moisture content with the highest volumetric moisture content in Zwevezele (526 kg m^{-3}) and the lowest in Wachtebeke (268 kg m^{-3}). There was no difference in OM content between the three types CFM at the start of the experiment. The OM content decreased for ST2 (relative decrease of 33%), SP2 (6%) and C2 (37%) in Wachtebeke and for C2 in Zoersel (24%). The CFM in Wachtebeke had a higher C/N ratio than did the CFM in Zwevezele at the start of the experiment. The C/N ratio decreased during the experiment for C2, ST2 and S2 in Wachtebeke and was lower for C2 and ST2 than for SP2. However, the C/N ratio did not change in the other locations. Total K content was lower for S2 compared to the other treatments in all locations. At the start of the experiment, there were no differences in mineral N concentrations among the three locations. During the experiment, the NO_3^--N concentration increased for the treatments in Wachtebeke and was higher for ST2 and C2 than for SP2. The NO_3^--N concentrations at the other locations were close to zero. The NH_4^+-N concentration was lower for C2 than for SP2 and S2 in Wachtebeke and did not decrease for any treatment in Zwevezele. The NO_3^--N/NH_4^+-N ratio was higher for C2 and ST2 than for SP2 in Wachtebeke. Those changes indicate a conversion of NH_4^+-N to NO_3^--N in C2 and ST2, indicating product stabilization. Only C2 in Wachtebeke had a NO_3^--N/NH_4^+-N ratio > 1, indicating a stable compost. There were no differences in OUR among the three types of CFM at the start of the experiment, but, according to the biodegradation potential, the CFM in Wachtebeke at the start of the experiment was more biodegradable than that in Zoersel and the CFM in Zwevezele had the lowest biodegradation potential at that moment. The OUR decreased for S2, SP2 and C2 in Wachtebeke, for ST2 and C2 in Zoersel and only for S2 in Zwevezele, indicating more stabilization. Moreover, all treatments in Wachtebeke and C2 in Zoersel showed the strongest decrease in biodegradation potential or the highest biodegradative activity. C2 in Wachtebeke and Zoersel and ST2 in Wachtebeke can be considered as ‘moderate stable’ according to Flemish compost standards (OUR between 10-

15 mmol kg^{-1} OM h^{-1}), this was confirmed by the lowest biodegradation potentials for C2 in Wachtebeke and Zoersel and ST2 in Wachtebeke (between 2.0 and 2.6).

From Experiment 2, we can conclude that the initial manure characteristics and storage conditions have an effect on the amount of N leaching and stability of the end product. In contrast to Experiment 1, composting resulted in higher soil mineral N concentrations than did uncovered stockpiling at all locations during Experiment 2. Higher soil mineral N concentrations under the piles were mainly the result of N leaching from the piles, but also higher soil temperatures had an indirect, albeit smaller effect. Generally, soil mineral NH_4^+-N amounts in the 0-10 cm layer were between 0.3 and 1.8% of the initial manure N content. Covering with TopTex or plastic did not influence the N leaching compared to uncovered stockpiling; however, covering with TopTex resulted in higher pile temperatures and a more stabilized end product.

Table 5.3 Average product quality of the cattle farmyard manure in Experiment 2 at the start (n = 4) and after storage without handling (S2), storage with a TopTex cover (ST2), storage with a plastic cover (SP2), and composting in combination with covering with TopTex cover (C2) (n = 3). Parameters at the start of the experiment, indicated with the same capital letter, are not significantly different over the locations (Scheffé test, $p < 0.05$). Parameters without an asterisk had a significant interaction term and were tested per location, treatments with the same letters are not significantly different (Scheffé test, $p < 0.05$). DM = dry matter, FM = fresh matter and OM = organic matter.

	Wachtebeke					Zoersel					Zwevezele				
	Day 1	Day 61				Day 1	Day 64				Day 1	Day 54			
		S2	ST2	SP2	C2		S2	ST2	SP2	C2		S2	ST2	SP2	C2
* Fresh bulk density (kg m^{-3})	347 A	513	485	398	507	599 B	652	634	592	642	656 B	757	715	675	718
** pH-H_2O (-)	8.5 A	8.9	9.0	9.0	9.1	8.8 B	8.5	8.7	8.5	8.8	8.7 AB	8.8	8.4	8.5	8.7
Electrical conductivity (µS cm^{-1})	1491 A	1619 a	2160 a	1646 a	1915 a	2593 B	1833 a	2587 b	2527 b	2897 b	2313 B	2797 a	2513 a	2397 a	2423 a
* Organic matter content (% of DM)	80.1 A	73.9	53.7	75.6	50.6	75.7 A	59.5	66.1	69.6	57.9	73.2 A	65.9	75.7	72.1	66.6
Dry matter content (% of FM)	22.8 A	20.2 a	25.4 ab	21.2 a	29.7 b	22.3 AB	22.6 a	24.6 a	23.3 a	25.5 a	19.9 B	21.9 a	19.1 b	19.2 b	20.2 ab
*** Volumetric moisture content (kg m^{-3} FM)	268 A	409 b	362 ab	314 a	356 ab	450 B	505 b	478 ab	454 a	479 ab	526 C	591 b	579 ab	545 a	573 ab
NO_3^--N (mg kg^{-1} DM)	2A	231 ab	1170 b	51 a	839 b	1 A	8 a	12 a	1 a	9 a	1 A	7 a	3 a	6 a	12 a
NH_4^+-N (mg kg^{-1} DM)	2727 A	2009 ab	1431 b	2003 b	491 a	3283 A	958 a	1378 a	1322 a	1860 a	4018 A	3339 a	3093 a	3784 a	3271 a
NO_3^--N / NH_4^+-N (-)	0.0007 A	0.144 ab	0.900 b	0.031 a	3.669 b	0.0003 A	0.008 a	0.008 a	0.001 a	0.005 a	0.0002 A	0.002 a	0.001 a	0.001 a	0.004 a
* N (g kg^{-1} DM)	26.1 A	28.1	26.2	25.7	26.0	33.0 A	25.5	27.8	29.8	28.2	27.3 A	26.1	27.3	27.6	25.9
* P (g kg^{-1} DM)	3.9 A	5.7	5.1	5.4	5.8	6.9 B	5.6	7.3	7.5	8.7	5.7 B	5.9	5.6	5.9	5.8
C/N (-)	17.1 A	14.7 b	11.5 a	16.5 b	10.9 a	13.0 AB	13.0 a	13.3 a	13.1 a	11.5 a	14.9 B	14.0 a	15.4 a	14.6 a	14.3 a
* C/P (-)	118 A	72	60	82	49	62 B	60	51	54	37	72 B	62	75	67	63
**** K (g kg^{-1} DM)	27.7 A	30.2 a	34.8 b	31.1 b	36.1 b	32.5 B	24.3 a	34.4 b	34.6 b	38.2 b	32.4 B	25.9 a	32.3 b	30.8 b	31.1 b
Ca (g kg^{-1} DM)	9.0 AB	13.7 a	14.3 a	11.6 a	15.4 a	7.6 A	6.7 a	8.7 a	10.0 a	14.8 b	11.2 B	11.8 a	10.5 a	10.8 a	10.9 a
Oxygen Uptake Rate (mmol kg^{-1} OM h^{-1})	31.2 A	17.7 a	10.5 a	24.7 a	15.6 a	39.3 A	35.1 a	18.6 b	23.5 a	13.9 b	43.0 A	19.8 a	32.4 a	29.6 a	20.2 a
Biodegradation potential (-)	7.8 C	3.6 b	2.6 a	4.2 b	2.0 a	5.4 B	4.6 c	4.1 b	4.2 bc	2.4 a	4.2 A	3.2 a	4.1 c	4.1 c	3.6 b

* no significant interaction, no effect of location and treatment

** no significant interaction, effect of location: pH in Wachtebeke is higher than in Zoersel and Zwevezele

*** no significant interaction, effect of location and treatment: lowest volumetric moisture content in Wachtebeke, highest in Zwevezele; lower volumetric moisture content for SP2 compared to S2 over the locations

**** no significant interaction, effect of treatment: lower K content for S2 compared to SP2, ST2 and C2 over the locations

5.3.3. Experiment 3: effect of adding bulking agent

In Experiment 3, we tested whether co-composting CFM with bulking agents could reduce the N leaching and improve the product quality.

5.3.3.1. Process monitoring

The mean ambient temperature and total precipitation during Experiment 3 were 12.7°C and 82.4 mm, respectively. The piles increased in temperature due to microbial activity directly after the start of the experiment. CG3 and CC3 continued to have high temperatures (> 50°C) during the entire experiment, while the temperature of C3 decreased slowly after turning on day 27 (Appendix Figure A7). After 56 days, CG3 and CC3 had temperatures of 59°C and 62°C, respectively, while C3 reached a temperature of 47°C.

5.3.3.2. N leaching

Soil characterization and temperatures

The soil temperatures (Appendix Figure A8) followed the same trend as the temperature in the piles. The soil temperature of C3 decreased after turning at day 27 and was lower than the soil temperatures of CG3 and CC3. During the first month, the soil temperatures of CC3 were higher than those of CG3. The maximum temperatures observed were 30°C, 34°C and 35°C for C3, CG3 and CC3, respectively.

N leachate losses under the piles

The total amount of N in the leachates collected in the containers under the piles was 0.01 ± 0 g, 0.31 ± 0.19 g and 8.61 ± 3.75 g per container for CC3, CG3 and C3, respectively. The mean leachate volume per container was 82 mL for CC3, 358 mL for CG3 and 3443 mL (full container so potentially even more) for C3.

Soil mineral N

At the end of the experiment, the soil NH_4^+-N concentrations (Figure 5.5A) under the piles were higher than the NO_3^--N concentrations (Figure 5.5B), and the NH_4^+-N concentrations were higher in the 0-10 cm layer than in the deeper soil layers. The NH_4^+-N concentration in the 0-10 and 10-30 cm layers was higher under C3 than in the other treatments and the reference soil layer without manure (Figure 5.5A). On a surface of 30 m^2, 3.83 kg NH_4^+-N could leach from C3 to the 0-90 cm layer (note that N mineralization due to higher soil temperatures could not be taken into account). This equals 4.2% of the initial N content of

the manure at the start of the experiment that was lost through leaching. For CC3 and CG3, this was only 1.7% of the initial N content. There was no difference in NH_4^+-N concentration among CC3, CG3 and the reference soil in the 0-10 and 10-30 cm layers. However, when considering the total mineral N concentrations in the 0-10 and 10-30 cm layers (Figure 5.5C), we found a higher concentration under CG3 and CC3 compared to the reference soil, but again, the soil mineral N concentration under C3 was significantly higher compared to the other treatments, indicating less leaching from CG3 and CC3 compared to C3. The NH_4^+-N concentration in the 0-30 cm layer under C3 was significantly higher (1231 kg ha^{-1}) than that in the reference soil and under CC3 (442 kg ha^{-1}) and CG3 (470 kg N ha^{-1}).

The NH_4^+-N concentrations at 30 cm from the piles (Figure 5.5A) were much lower than those under the piles; they were of the same order of magnitude as the NO_3^--N concentrations (Figure 5.5B). There were no differences in NH_4^+-N concentrations next to the piles among the different treatments.

5.3.3.3. Agricultural value of the products

Table 5.4 summarizes the results of the analyzed parameters of the feedstock materials and the end products of C3, CG3 and CC3.

At the start of the experiment the OM and DM contents of the CFM was 59.2% of DM and 25.8% of FM, respectively. By adding hay of grass and pre-composted hay to the CFM, drier products with a higher OM content, C/N and C/P ratio were added. At the end of the process, no significant differences were found between the OM content of the treatments, indicating a stronger OM degradation for CC3 and CG3. The DM content of CC3 was higher than CG3 and C3, while CG3 was drier than C3. The fresh bulk density and volumetric moisture content was different among the treatments, being highest for C3, followed by CG3 and C3. After 56 days, the NO_3^--N concentration was higher for CG3 than for C3 and CC3, while there were no differences in NH_4^+-N concentrations. Furthermore, the total N, K and P concentrations were higher for C3 and CG3 compared to CC3, due to the lower nutrient concentrations of the pre-composted hay. The NO_3^--N/NH_4^+-N ratios were lower than 1 for all treatments, indicating that composts were not yet stabilized. The OUR of C3 (13.5) was higher than that of CG3 (6.9), the OUR of CC3 was in between (10.1). At the end of the experiment, the percentage cellulose and lignin was lower for C3 than for CG3 and CC3, resulting in a higher biodegradation potential for C3. Consequently, C3 was less stable

compared to CG3 and CC3 according to OUR and biodegradation potential. Less than two emerging weeds were detected over the different treatments.

From Experiment 3, we can conclude that co-composting CFM with bulking agents reduced the soil mineral N concentrations under the piles and improved the product stability, compared with composting only CFM.

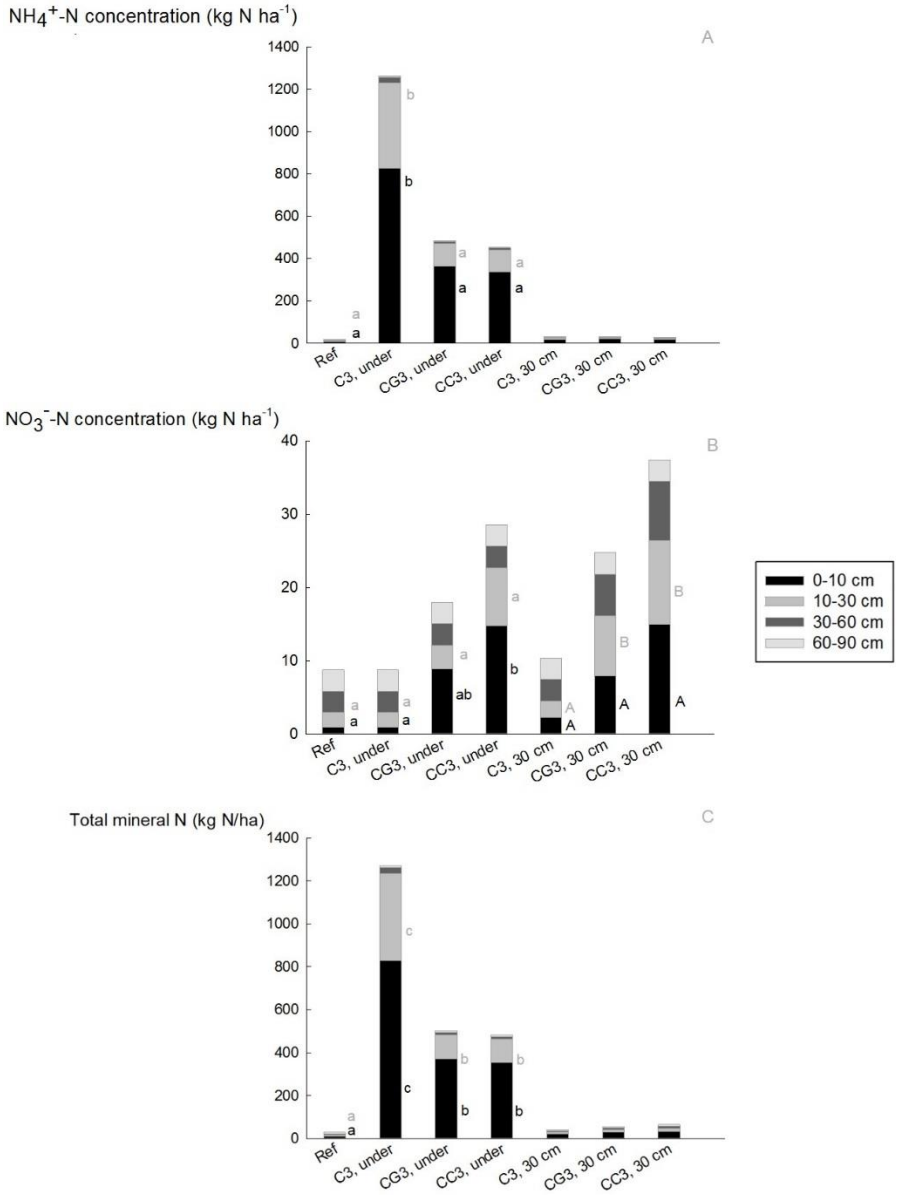

Figure 5.5 (A) Soil NH_4^+-N, (B) NO_3^--N concentrations and (C) total mineral N (NH_4^+-N + NO_3^--N) concentrations (n = 3) in the 0-10, 10-30, 30-60 and 60-90 cm soil layers under the piles and at 30 cm from the piles (just outside the TopTex covers) at the end of Experiment 3 for a reference situation (soil not covered by a manure pile) and for the cattle farmyard manure that was composted (C3), co-composted with hay of grass (CG3) and co-composted with pre-composted hay of grass (CC3). Treatments with the same letter are not significantly different (Scheffé test for 0-10 and 10-30 cm layers, small letters: comparison between treatments under the pile, capital letters: comparison between treatments at 30 cm from the piles, $p < 0.05$).

Table 5.4 Feedstock (cattle farmyard manure, hay of grass and pre-composted hay of grass) characterization at the start of Experiment 3, and product quality after 2 months for cattle farmyard manure that was composted (C3), co-composted with hay of grass (CG3) and co-composted with pre-composted hay of grass (CC3) (mean ± standard deviation; $n = 4$). Parameters at the end of the experiment with the same letters are not significantly different (Scheffé test, $p < 0.05$). DM = dry matter, FM = fresh matter and OM = organic matter, nda = no data available.

	Day 1 (feedstock materials)			Day 56					
	Cattle manure	Old grass	Pre-composted grass	C3		CG3		CC3	
Fresh bulk density (kg m^{-3})	399 ± 69	132 ± 29	170 ± 16	641 ± 21	c	391 ± 27	b	330 ± 18	a
pH-H_2O (-)	8.3 ± 0.4	nda	7.0 ± 0.2	9.1 ± 0.2	b	8.8 ± 0	ab	8.6 ± 0.1	a
Electrical conductivity (µS cm^{-1})	1197 ± 36	nda	196 ± 32	2219 ± 205	b	1625 ± 119	a	1328 ± 79	a
Organic matter content (% DM^{-1})	59.2 ± 11.6	84.3 ± 3	62.8 ± 3.2	51.6 ± 4.9	a	53.4 ± 3.9	a	50.4 ± 3.2	a
Dry matter content (% FM^{-1})	25.8 ± 6.8	32 ± 6.7	39.4 ± 1.7	28.2 ± 2.2	a	38.3 ± 3.9	b	49.3 ± 0.9	c
Volumetric moisture content (kg m^{-3} FM)	299 ± 75	90 ± 24	103 ± 10	460 ± 18	c	242 ± 28	b	167 ± 9	a
NO_3^--N (mg kg^{-1} DM)	12 ± 13	nda	75 ± 7	43 ± 33	a	169 ± 85	b	93 ± 27	ab
NH_4^+-N (mg kg^{-1} DM)	1641 ± 496	nda	75 ± 7	4326 ± 2367	a	1823 ± 2155	a	702 ± 71	a
NO_3^--N / NH_4^+-N (-)	0.007	nda	1	0.008 ± 0.001	a	0.122 ± 0.029	a	0.133 ± 0.028	a
N (g kg^{-1} DM)	19.9 ± 6.9	24.8 ± 4.2	14.1 ± 1.3	26.8 ± 2.7	b	26.2 ± 3.2	b	20.6 ± 1.3	a
P (g kg^{-1} DM)	3.0 ± 0.6	3.2 ± 1.2	1.6 ± 0	4.8 ± 0.5	b	4.4 ± 0.3	b	3.2 ± 0.1	a
C/N (-)	17.3 ± 3.8	19.3 ± 3.4	24.9 ± 1.9	10.7 ± 1	a	11.5 ± 1.6	ab	13.6 ± 1.1	b
C/P (-)	109 ± 10	164 ± 61	225 ± 14	60 ± 6	b	68 ± 5	a	87 ± 6	a
K (g kg^{-1} DM)	nda	17.0 ± 7.2	7.7 ± 0.3	23.1 ± 2.1	a	22 ± 0.4	a	18.3 ± 0.8	b
Ca (g kg^{-1} DM)	nda	9.3 ± 2.2	6.7 ± 0.3	15.8 ± 2.0	b	13.1 ± 1.0	ab	11.5 ± 0.5	a
Oxygen Uptake Rate (mmol kg^{-1} OM h^{-1})	nda	nda	8.9 ± 2.2	13.5 ± 2.8	b	6.9 ± 1.0	a	10.1 ± 2.4	ab
Hemicellulose (% OM^{-1})	nda	21.7 ± 4.5	18.6 ± 3.8	20.9 ± 2.4	a	21.4 ± 3.6	a	18.8 ± 4.8	a
Cellulose (% OM^{-1})	nda	20.5 ± 1.5	22.2 ± 2.1	17.6 ± 1.8	a	20.9 ± 1.6	b	25.0 ± 2.6	c
Lignin (% OM^{-1})	nda	19.0 ± 1.4	15.9 ± 0.5	21.5 ± 0.9	a	26.6 ± 0.4	b	28.8 ± 1.0	c
Biodegradation potential (-)	5.1 ± 1.1	2.2 ± 0.3	2.6 ± 0.2	1.8 ± 0.3	b	1.6 ± 0.4	a	1.5 ± 0.3	a
Germinated weeds (amount L^{-1})	nda	nda	nda	1 ± 1	a	2 ± 0	a	2 ± 1	a

5.4. Discussion

5.4.1. Process and product quality

The storage method (uncovered stockpiling, covering with plastic or geotextile, composting) clearly influenced the temperature development in the piles. In Experiment 2, composting resulted in higher pile temperatures than with uncovered stockpiling, related to the higher microbial activity in the compost piles by turning (El Kader et al., 2007; Brito et al., 2008; Parkinson et al., 2004). However, in Experiment 1, uncovered stockpiling led to higher temperatures than did composting. A potential explanation is that, in Experiment 1, contrary to Experiment 2, the CFM was loosened and homogenized by a manure spreader before it was put on the field, hence starting the composting process (Godden and Penninckx, 1997) and leading to a similar temperature development for both treatments in the beginning. However, in the composted CFM, the infiltration of rainwater was prevented by covering, and extra water was lost by turning, resulting in drier circumstances compared to the uncovered stockpiled manure, probably slowing down the degradation process and, thus, the pile temperature. Since an impermeable plastic cover restricts aeration, OM degradation and internal heat production, semi-anaerobic storage of CFM led to a gradual decrease in pile

temperature from the start and resulted in lower temperatures than did uncovered stockpiling, as seen in previous research (Chadwick, 2005; Brito et al., 2008; Shah et al., 2012). Covering with a geotextile resulted in higher pile temperatures than in uncovered stockpiling during winter because the cover has an isolating effect. Piles covered with a geotextile had higher temperatures than did piles covered with a plastic cover, because the geotextile allows aeration and a higher microbial activity; as was also found by Brito et al. (2008) for the solid fraction of cattle slurry.

Besides the storage method, also the initial characteristics of the CFM influenced the pile temperature. For example, in Wachtebeke temperatures and biodegradative activity were generally higher than in Zoersel and Zwevezele, independent of the storage method; also, adding hay or precomposted hay as bulking agent increased the pile temperatures compared with only composting CFM, since the added hay improved aeration in those piles (Magri and Rosa Teira-Esmatges, 2015). Multiple linear regression of the three experiments illustrated that the average pile temperature was negatively correlated with the initial volumetric moisture content of the CFM ($R^2 = 0.70$, $p < 0.001$, $F = 31.86$). In conclusion, drier CFM with a lower bulk density, i.e., CFM with more straw or with additional bulking agents, led to higher pile temperatures during storage. High pile temperatures (> 55°C for 15 days) result in mortality of plant pathogens and emerging weeds, as also indicated by the absence of germinated weeds in the end products. However, only both treatments in Merelbeke and the compost treatments in Wachtebeke and Diest reached sufficiently high temperatures for 15 days.

The pile temperatures were, thus, influenced by the storage method and initial manure characteristics. Moreover, the pile temperatures had an effect on the product stability, since the OUR, volumetric moisture content and OM content of the CFM end products were negatively correlated to the average pile temperatures ($R^2 = 0.84$, $p < 0.001$, $F = 28.3$). In other words, it was shown that the higher the pile temperatures were in the present study, the better the decomposition and stabilization of the CFM was (lower OUR, volumetric moisture content and OM content). In Experiment 2, only composting in Wachtebeke and Zoersel and covering with a geotextile in Wachtebeke resulted in stable end products (OUR < 15) due to the higher pile temperatures. In Zwevezele, the differences among the end products were small due to the overall low pile temperatures. In Diest, all composts were stable, but co-composting increased the stability, as illustrated by an OUR < 10 and lower biodegradation potential. Additional advantages of composting compared with covering are the mass and

volume reductions (Breitenbeck and Schellinger, 2004), homogenization and the destruction of pathogens (Lung et al., 2001) and weed seeds (Eghball and Lesoing, 2000).

5.4.2. N leaching

In all experiments, storing CFM on the field increased the concentrations of soil mineral N and especially the NH_4^+-N concentrations under the piles. According to Dewes et al. (1993) and Parkinson et al. (2004) NH_4^+-N is the most important N compound lost by leaching during composting and the loss of NO_3^--N is negligible. Due to limited oxygen diffusion under the manure the conversion from NH_4^+-N to NO_3^--N was prevented. Soil mineral N concentrations were higher in the top soil layer compared to the deeper soil layers because NH_4^+-N is less mobile than NO_3^--N. Over the different treatments, the NH_4^+-N concentrations under the piles in the 0-90 cm layer was between 0.3 and 4.2% of the initial manure N content, which is similar to the results of Petersen et al. (1998), Sommer and Dahl (1999) and Parkinson (2000) who reported losses between 0.5 and 2.5% of the initial N content by leaching. Those losses are relatively low compared to volatile N losses; for example, Martins and Dewes (1992) and Parkinson (2000) reported volatile losses between 11-51% of the initial N content of the cattle manure. Furthermore, it should be noted that the reported N concentrations under the piles between 200-1231 kg ha^{-1} seem very high, but are local and limited to the soil under the pile. NH_4^+-N is less prone to leaching, but after removal of the manure, the soil may become more aerobic and NO_3^--N might be formed, with a higher risk for leaching (Peigne and Girardin, 2004). As CFM is stored on field during winter, NO_3^--N can be used by the following crop; consequently, less fertilization will be needed on the location of the piles. Furthermore, the risk for leaching can be mitigated, for example, by annually alternating the location of the manure piles, with sufficient distance from vulnerable zones such as watercourses. We could distinguish between two different processes resulting in a higher mineral N content under the piles, namely, (1) direct leaching of mineral N from the CFM to the soil and (2) higher soil temperatures under the piles (up to 37°C) resulting in both N mineralization of organic N from the leachate and soil organic N in the top soil.

It remains difficult to explain the observed variation in N leaching among the different treatments, although we clearly observed that it was dependent on a combination of the initial manure characteristics and the storage option. In contrast to Experiment 1, a higher concentration of NH_4^+-N was observed in the 0-30 cm soil layer under the compost piles than in the uncovered stockpiled manure in Experiment 2. Composting manure with an initially high volumetric moisture content and low C/N ratio (as in Experiment 2) resulted

in higher temperatures and, therefore, possibly extra N mineralization in the pile, and thus more N in the leachate than under stockpiling. Moreover, turning the pile after one month could have increased the leachate losses from the pile to the soil (Parkinson, 2000). This result was similar to those of the study of Parkinson et al. (2004) who also found lower mineral N losses in manure stacks than in turned treatments. Composting and turning manure with an initially low volumetric moisture content and high C/N ratio (as in Experiment 1) led to further drying of the manure, which could have limited leaching. Furthermore, uncovered stockpiling of this type of manure resulted in a higher moisture availability than did composting, which could have increased microbial activity and N mineralization in the pile, as higher temperatures were observed during stockpiling than during composting. During storage, the mineralized N could leach through the infiltration of rainwater in the pile. Experiment 3 showed that the volumetric moisture content and C/N ratio of the feedstock mixture improved by adding bulking agents, resulting in a better composting process and a more stabilized end product than after only composting CFM. Of course, by adding those bulking agents, the CFM was only half of the pile volume. Nevertheless, little leachate and soil NH_4^+-N under the co-composted piles was noticed, as was also found by Ulen (1993). However, composting CFM requires a change in manure composition and an appropriate monitoring, as the manure is put on windrows and is turned so as to aerate the piles, which is more labor-intensive. Remarkably the soil N concentrations were higher in Experiment 3 compared with the other experiments. This could be due to the fact that the piles were turned once more, since there is a close relationship between higher concentrations of nutrients in leachate and increased number of turning events (Parkinson, 2000).

There was no difference in N leaching under the piles between uncovered storage and covering with plastic or geotextile for the wet manure with low C/N ratio (Experiment 2). This was in contrast to other literature (Ulen, 1993) where leachate and volatile losses are reduced by covering since rainwater cannot enter the piles. However, in the latter study, drier manure with a higher C/N ratio (similar to Experiment 1) was tested. Covering with plastic or geotextile resulted in lower soil NH_4^+-N concentrations next to the piles (outside the cover) than when no cover was used, because the run-off water was not in contact with the manure (Ulen, 1993).

Our results indicate that regional regulations on manure application can be optimized. In comparison with the surrounding regions, there are clear differences in legislation. In the

Walloon region of Belgium, farmyard manure with sufficient straw content (depending on specific stable conditions) and compost with a minimal DM content of 35% can be stored on the field for a maximum of 10 months, under certain conditions such as a minimal distance of 20 m from watercourses and not on a slope of more than 10%. In the Netherlands, if the storage period is less than six months, it is sufficient to store farmyard manure 5 m from a vulnerable zone, under a cover and on an absorbing layer of a minimum of 15 cm and 25% organic material (e.g., straw). Only when the storage period is longer than six months, it is necessary to put the farmyard manure on a concrete floor including the collection of rainwater and leachate.

5.5. Conclusion

In this study, different storage options for CFM were evaluated so as to minimize N losses and groundwater contamination, while improving the agronomic quality since CFM is a valuable organic fertilizer. Field storage of CFM resulted in higher soil NH_4^+-N concentrations under the piles than in the reference soil without stored manure. In the 0-90 cm soil layer, those soil NH_4^+-N concentrations reached max. 4.2% of the initial manure N content, which is low compared with reported volatile N losses. The major cause was direct N leaching from the pile; however, elevated soil temperatures under the piles had an indirect effect on extra N mineralization of organic N in the leachate and soil organic N. The storage conditions and initial CFM characteristics determined the N losses to the soil, but also the stability of the end product since the initial volumetric moisture content of the CFM was negatively correlated with the average pile temperature. With drier CFM of a lower bulk density and a high C/N ratio, i.e., CFM with more straw or with additional bulking agents, composting led to a more stable end product and less N leaching. With wetter CFM of a higher bulk density and a low C/N ratio, thus with low amounts of straw, stockpiling and covering (plastic or geotextile) resulted in the least N leaching to the soil. When covering the CFM with a geotextile, the most stable end product was achieved, because this allowed aeration and thus a higher pile temperature. The present study was the first to give an indication about the N leachate losses and product quality for different treatment options of different types of CFM at a field scale. However, more research is necessary, focusing on the evaluation of N losses, both liquid and volatile, over the whole manure management chain, from stable, over storage, spreading on the field and further degradation in the soil.

CHAPTER 6

Improving the product stability and fertilizer value of cattle slurry solid fraction

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Abstract

Separating dairy cattle slurry in a liquid and solid fraction (SF) is gaining more interest, since it enables a more targeted use of both fractions. However, the valorization of the SF is limited on P-rich soils, due to its high P content, and the export or use as bedding material requires sanitation. Therefore, we investigated the influence of composting or ensiling the SF, whether or not mixed with bulking agents, on the product quality in terms of fertilizer value, sanitation and stability. Ensiling can be considered as a controlled storage method for conserving C and nutrients. Soil amendment with co-ensiled SF resulted in a higher N mineralization and crop growth compared to amendment of co-composted SF. Co-composting SF with structure-rich feedstock materials optimized the composting process and sanitation when compared with composting pure SF and did not increase the risk for X-TAS spores. Further, the composts contained more P per unit of fresh weight than the silages, beneficial for the export of the composted SF. The oxygen uptake rate was found to be less powerful to determine the stability of fresh, composted and ensiled SF.

6.1. Introduction

Inappropriate application of cattle slurry can cause severe environmental problems (e.g. input of harmful trace metals, inorganic salts and pathogens, nutrient leaching and emissions of toxic gases), especially in regions where the amount of cattle manure exceeds the loading capacity of soils available for manure application (Gomez-Brandon et al., 2008; Hutchison et al., 2005). Separating liquid and solid fractions of cattle slurry is gaining more interest, since it enables a more targeted use of both fractions: the liquid fraction is less rich in P compared to the solid fraction (SF) (Ford and Fleming, 2002), hence rebalancing the N/P ratio enables applications of this liquid phase which better suit crop requirements. According to Schröder et al. (2009) N/P ratio equals 7.6, 5.9 and 8.0 for slurry, SF and liquid fraction, respectively. Furthermore, due to the increased N/P ratio of the liquid fraction, higher doses could be applied per hectare. The direct use of the SF as a fertilizer is then limited, particularly given the restrictions in P fertilizer application in soils (e.g. Manure Decree in Flanders) due to the EU Nitrates and Water Framework Directive. On the other hand, the use of SF (directly or after treatment) can be a cheap and valuable means to maintain soil organic carbon levels (Vanden Nest et al., 2016). The latter is a major challenge since the soil organic carbon content of many croplands in temperate regions is declining. Besides the potential use as soil improver, the SF could be exported and used as a crop fertilizer abroad or could be used as bedding material in stables. In both cases sanitation is a key requirement. The implementation in Flanders of the EU regulation (EC 1069/2009 of the European Parliament and of the Council of 21 October 2009) requires heating the material at least 1 hour above 70°C as well as a bacteriological analysis before export. Use of SF as bedding material is currently not allowed in Flanders. In contrast, in the Netherlands, where the EU regulation is interpreted differently, use of SF as bedding material is allowed on the farm where the cattle slurry is produced. Dutch dairy companies discourage the use of compost and composted organic materials as bedding material, since the transfer of spores of thermophilic aerobic spore-forming bacteria with an exceptionally high heat resistance (X-TAS) from these types of bedding to raw milk is assumed to pose a risk for the quality of the milk and milk products, in particular for the shelf-life of sterilized milk products (Driehuis et al., 2014).

Processing the SF before use may alleviate this problem. Two promising ways of treating SF are (1) composting and (2) ensiling, both of which are studied in this paper. Controlled composting of solid manure has several advantages, such as the potential destruction of weed

seeds and pathogens, homogenization and reduction in mass and moisture content which makes the manure easier to store, transport and spread, and manure stabilization which prevents negative impacts on plant growth (Bernal et al., 2009). A potential disadvantage is the high risk for N losses during composting (Eghball et al., 1997), not only posing an environmental problem but also reducing the amount of available N for plant growth, i.e. the nutrient use efficiency of the SF. Composting animal manure with a high moisture content and limited free air space is problematic (Viaene et al., 2016a). Brito et al. (2008) reported that the efficiency of the composting process and OM stability were improved by increasing the DM content of SF. Thereto, C-rich bulking agents can be added in order to compost cattle (Aguerre et al., 2012; Michel et al., 2004) or pig (Nolan et al., 2011) slurry, and SF after separating digestate (Bustamante et al., 2012). Hence, we investigated the potential of co-composting SF with bulking agents. Thereby, the use of on-farm available byproducts as bulking agents, such as grass clippings, straw, and straw-rich cattle manure, is supposed to be straightforward, cheap and efficient. Using additives in the composting process could be another option to reduce gaseous losses and improve the agronomic value of the end product. Clinoptilolite, the most common natural zeolite in the world, has a high cation exchange capacity especially for NH_4^+-N (Hedstrom, 2001). Adding clinoptilolite (6.25% on fresh weight basis) to dairy slurry reduced ammonia gas with 50% by adsorption of NH_4^+-N (Lefcourt and Meisinger, 2001). Furthermore, Shah et al. (2012) demonstrated that clinoptilolite reduced N loss during and after storage of cattle manure in the stable.

Ensiling is a widespread technique for storing many field crops (e.g. grass and maize), thereby producing fodder or feedstock for anaerobic digesters. This technique can be applied for manure as well (Viaene et al., 2016a). According to Thomsen and Olesen (2000b), anaerobic storage is superior to composting when considering the manure as a N resource, since anaerobically stored manure contains more mineral N than composted manure. Viaene et al. (2016a) concluded that semi-anaerobic storage of cattle farmyard manure resulted in lower pile temperatures and a less stabilized end product with higher mineral N concentrations compared to composting. Thus, ensiling could be an alternative management option for the SF, however, given the high moisture content of the SF, it is presumable that, comparable to composting SF, bulking agents should be added.

The effect of adding bulking agents or clinoptilolite to the SF of cattle slurry in an aerated on-farm windrow composting has received little attention, and to the best of our knowledge

(co-)ensiling SF of cattle slurry has not been investigated previously. Therefore, the **first aim** of this study was to test the following hypotheses:

1. Ensiling is a controlled storage method resulting in a non-stabilized end product that further decomposes after soil addition. Ensiling the SF could optimize the fertilizer value of the SF in terms of conservation of C and nutrients.
2. Composting generally results in a more stable and sanitized product. The addition of bulking agents, such as straw-rich cattle manure or a mixture of straw and fresh grass clippings, will enhance the composting process and thereby optimize the agronomic value of the end product in terms of sanitation and stability, both important criteria for export.
3. Composting as well as co-composting of the SF does not increase the concentration of X-TAS spores, important for use as bedding material.
4. Adding clinoptilolite to the SF reduces N losses during composting and thus reduces the negative environmental impact. This conservation of N from the initial SF is also in favor of the fertilizer value of the end product.

More specifically, the difference in process, fertilizer value (in terms of conservation of nutrients and effect on N uptake in perennial ryegrass) and product stability among the treatments were compared. The product stability is an important factor to assess the fertilizer value, as application of unstable products can immobilize N from the soil, resulting in a negative effect on crop growth (Bernal et al., 2009). Stability is strongly related to the rate of microbial activity in the compost (Gomez-Brandon et al., 2008), however, there are many indicators to determine this and it cannot be established by one single parameter (Bernal et al., 1998; Bernal et al., 2009). For animal manure-based composts, stability is evaluated by e.g. a decrease in pile temperature to ambient air temperature (Brito et al., 2012), a decrease in organic matter (OM) content and C/N ratio to approximately 20:1 (Larney and Hao, 2007), a lower oxygen uptake rate (OUR) (Bernal et al., 2009), an NH_4^+-N concentration lower than 400 mg kg^{-1} DM and the conversion of NH_4^+-N to NO_3^--N resulting in a NO_3^--N/NH_4^+-N ratio > 1 (Zucconi and de Bertoldi, 1987). The biochemical composition (relative amounts of cellulose, hemicellulose and lignin) of the end products are also determining the stability of the OM (Lashermes et al., 2012; Veeken et al., 2007). However, few studies have used cell wall components as a successful stability parameter for manure-containing composts and silages. Therefore, the **second aim** of this study was to compare different stability indicators for composted and ensiled SF.

6.2. Materials and Methods

6.2.1. Treatments and process monitoring

The experiments ran for two months (mid-April to mid-June 2014) and were conducted in an open-air composting facility with a concrete pad at the Institute of Agricultural and Fisheries Research (ILVO), Merelbeke, Belgium. Shortly before the start, fresh dairy cattle slurry (animal feed: 60% maize silage, 15% grass silage and 15% beet pulp) was separated with a mobile screwpress (type XXXL, Maverko, The Netherlands) and was covered during storage (10 days) until the start of the experiment. At the start of the experiment, a windrow with a volume of approximately 24 m³ (8 m length x 3 m width x 1 m height) was set up for each of the five treatments:

- composting pure SF (24 m³) (*SF_C*)
- composting SF (24 m³) with 2% clinoptilolite (Orffa, Bornem, Belgium) on dry weight (*SF+Clin_C*)
- co-composting SF (12 m³) with straw-rich cattle farmyard manure (CFM) (12 m³) (*SF+CFM_C*)
- co-composting SF (16 m³) with straw (4 m³) and fresh grass clippings (4 m³) (*SF+S+G_C*)
- co-ensiling SF (16 m³) with straw (4 m³) and fresh grass clippings (4 m³) (*SF+S+G_E*)

The CFM, straw and fresh grass clippings consisted, respectively, of 83%, 95% and 89% OM on dry matter (DM); 32%, 86% and 23% DM; 18.4, 4.9 and 34.7 g N kg^{-1} DM and 3.3, 1.2 and 3.7 g P kg^{-1} DM. The total amount of feedstock per treatment was weighed before and after the experiment. Effective Micro-organisms (EM; Agriton, Mesen, Belgium) were added at a rate of 2 L m^{-3} to the silage treatment. The compost piles were covered with a geotextile (TopTex®, TenCate, Almelo, The Netherlands) which hinders the infiltration of rainwater and allows gas exchange, while the silage treatment was covered and sealed with a plastic cover to ensure anaerobic conditions. Temperature and CO_2 levels in the compost piles were monitored manually 3-4 times per week on four different points in the middle of the pile. Similar, for the silage, temperature was measured on two different points (the holes were recovered with plastic tape after the measurement), while CO_2 was measured at the end of the experiment. Compost piles were aerated when temperatures exceeded 65°C or CO_2 levels exceeded 16%, using a compost turner (TG 301, Gujer Innotec AG, Mesikon,

Switzerland). Daily average temperature and precipitation data were collected from the weather station located near the composting facility.

Additionally, to confirm the results of the silage treatment on field scale (SF+S+G_E), simultaneous silage experiments were conducted on lab scale, using ensiling buckets of 15 L (Agriton) for all treatments in four replicates:

- ensiling pure SF (*SF_E*$_{lab}$)
- ensiling SF with 2% clinoptilolite on dry weight (*SF+Clin_E*$_{lab}$)
- co-ensiling SF with straw-rich CFM (*SF+CFM_E*$_{lab}$)
- co-ensiling SF with straw and fresh grass clippings (*SF+S+G_E*$_{lab}$)

The ensiling buckets were filled with the same feedstock composition (on volume basis) as in the compost experiment. The buckets were closed airtight to ensure anaerobic conditions and contained a reservoir to collect possible leachate from the silages. The material loss was calculated by weighing the buckets at the start and the end. Furthermore, silage temperature was measured at the end of the experiment.

6.2.2. Product characterization and stability

The different feedstock materials and the feedstock mixture were characterized at the start of the experiment. To analyze the product quality, samples were taken at day 32 and at the end of the experiment (day 61). Sampling was performed by collecting 10 subsamples of ± 3 L each over the whole length of the pile; after thoroughly mixing in a container of ca. 0.8 m^3, a sample was taken and this was repeated four times (= four mixed samples per material). Samples were analyzed for physico-chemical compost quality parameters: DM content (EN 13040), OM content (EN 13039), pH-H_2O (EN 13037), electrical conductivity (EC) (EN 13038), mineral N content (NH_4^+-N and NO_3^--N, ISO 14256-2) and total N content (Dumas EN 13654-2). Total Ca, K, Mg and Na concentrations were measured by a charge-coupled device (CCD) simultaneous Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES; VISTA-PRO, Varian, Palo Alto, CA), after ashing and digestion of the samples with 7N HNO_3. Total P was measured in the same extract with a Varian CARY 50 Spectrophotometer. For determination of the number of emerging weeds, 500 mL product was mixed with 2000 mL white peat and spread in a layer of 2-3 cm in a container, and kept for 3 weeks at 21°C and 100% relative humidity under conditions of sufficient natural light. The number of emerging weeds were counted after three weeks. As an indicator of compost stability, OUR was calculated from the oxygen consumption due to microbial activity of 20

g compost (< 1 cm fraction) in 200 mL buffered nutrient solution (with N-allylthiourea (6.25 mg per flask) as nitrification inhibitor and a phosphate buffer to buffer the suspension at pH 6.5) in a l L Schott flask during five days of shaking at 120 rpm in a closed OxiTop respirometer at 20°C based on the method reported in Grigatti et al. (2011). OUR was expressed as mmol kg^{-1} OM h^{-1}. As demonstrated before (Hutchinson and Griffin, 2008; Lashermes et al., 2012), the measurement of cell wall components according to the method of Van Soest et al. (1991) is another method to estimate the product stability. The biodegradation potential can then be estimated by the (hemicellulose+cellulose)/lignin ratio with hemicellulose = neutral detergent fiber (NDF) – acid detergent fiber (ADF), lignin = acid detergent lignin (ADL) and cellulose = ADF-ADL (Van Soest et al., 1991). The cell wall components were expressed as a percentage on OM content. For the ensilage treatments at lab scale, one mixed sample per bucket was characterized (same parameters as described above).

6.2.3. Determination of spores

At the start of the experiment and after 61 days the products of SF_C, SF+S+G_C were analyzed for populations of spores of thermophilic aerobic spore-forming bacteria with different heat resistance properties at NIZO food research B.V. (The Netherlands). To compare the results, CFM and composted CFM (CFM_C) were analyzed as well. Spore counts were determined in extracts of triplicate samples. Volumes of 5 mL of sample extract were heated in a water bath for 10 min at 80°C, in boiling water for 30 min at 100°C and in an oil bath for 20 min at 115°C. After cooling down, serial dilutions in PSS were made. Spores of thermophilic spore-forming bacteria were enumerated on double-layered pour plates of tryptone soy agar containing 0.2% soluble starch, incubated for 2-4 days at 55°C. Spore counts after the heat treatments of 10 min at 80°C and 30 min at 100°C were determined after two days incubation, spore counts after the heat treatment of 20 min at 115°C were determined after four days incubation. These populations were designated as low-heat-resistant spores of thermophilic aerobic spore-forming bacteria (L-TAS), high-heat-resistant spores of thermophilic aerobic spore-forming bacteria (H-TAS) and extreme-heat-resistant spores of thermophilic aerobic spore-forming bacteria (X-TAS), respectively.

6.2.4. Soil characterization for N mineralization and N fertilizer replacement value experiment

A sandy loam (USDA) soil, containing 58.1% sand (50 - 2000 µm), 35.3% silt (2-50 µm) and 6.7% clay (< 2 µm), was air-dried and sieved on a 2 mm mesh for the N mineralization experiment (in which small volumes of soil were used), and on a 2.5 cm mesh for the nitrogen fertilizer replacement value (NFRV) experiment (in which larger volumes of soil were used) to obtain a more realistic approximation of field conditions. The soil characteristics prior to the experiment (n = 3) were pH-KCl = 6.23 ± 0.02; TOC = 0.90 ± 0.02%; hot water extractable C = 1163 ± 49 mg kg^{-1} DM; NH_4^+-N = 1.4 ± 0 kg ha^{-1} ; NO_3^--N = 2.6 ± 0 kg ha^{-1}; P-$CaCl_2$ = 4.0 ± 0.1 mg kg^{-1} DM and P-AL = 226 ± 4 mg kg^{-1} DM.

6.2.5. Nitrogen mineralization

A N incubation experiment was set up on September 15th 2014 with the end products (products were stored at < 4°C until the incubation trial was started) of the following treatments: SF_C, SF+Clin_C, SF+S+G_C and SF+S+G_E. Each treatment consisted of three replicates. The products were added to the soil based on an equal N input of 170 kg N ha^{-1}, which is the maximum fertilization dose per ha per year with animal manure in Flanders (Flemish Land Agency, 2015). An extra treatment with additional N (60 kg N ha^{-1}) was added for the ensilaged SF with straw and grass (SF+S+G_E_{60N}), because it was hypothesized that the ensilaged product could temporarily immobilize N. Furthermore, two control treatments without organic amendments were included: one with (CON_{60N}) and one without (CON) application of N fertilizer. N fertilizer was applied at a rate of 60 kg N ha^{-1} (treatments CON_{60N} and SF+S+G_E_{60N}) as NH_4NO_3 (35% N, Merck) that was dissolved in demineralized water prior to addition. Products of SF+S+G_C and SF+S+G_E were manually cut into pieces smaller than 2 cm^2 prior to application to the soil. The soil was thoroughly mixed with the products and demineralized water (to obtain a moisture content of 50% water filled pore space (WFPS)), and subsequently placed in PVC tubes (h = 12 cm, r = 2.3 cm). The mixture was then gently manually compacted to obtain a bulk density of 1.4 g cm^{-3}. The tubes were covered with a single layer of gas permeable Parafilm® (MBarrier Film (Pechiney Plastic Packaging, Chicago, IL, USA) to avoid water loss, allowing aeration, and subsequently incubated at 15°C and 70% relative humidity. The tubes were weighed frequently and demineralized water was added when needed to maintain a WFPS of 50%. Soil was sampled destructively by removing intact tubes, and analyzed for mineral N and

pH-KCl at the start of the incubation and on day 18, 36, 53, 71 and 99 (methods in section 6.2.2). The N mineralization (N_{min}) on each sampling date was calculated by subtracting $N_{min, control}$ from $N_{min, product}$ with $N_{min, product}$ the amount of mineral N released in the soil amended with the different treatments of SF, with or without N fertilizer, and $N_{min, control}$ the amount of N released in the control treatment without organic amendment, with or without N fertilizer. We assumed a linear mineralization to compare the different products: a linear regression was performed where N_{min} = a*time + b, with N_{min} relative to the N input by the products and/or fertilizer dose (%) and time in days.

6.2.6. Nitrogen fertilizer replacement value

The pot experiment with perennial rye grass (*Lolium perenne* L., Melpetria tetra) took place in a greenhouse (average temperature of 20°C) with assimilation lights (from 5 am until 10 pm) to determine the nitrogen fertilizer replacement value (NFRV) of the end products (products were stored at < 4°C until the pot trial was started) of SF_C, SF+Clin_C, SF+S+G_C and SF+S+G_E. Additionally, N fertilizer was applied at four different rates to the control soil (CON) to obtain N response curves necessary to calculate NFRV (CON_{0N}, CON_{50N}, CON_{100N}, CON_{150N}). Each treatment consisted of three replicates. The products were added based on an equal N input of 100 kg N ha^{-1}, which is below the maximum N dose with animal manure of 170 kg N ha^{-1} $year^{-1}$ (Flemish Land Agency, 2015). To supply sufficient P, P_2O_5 was added (triple superphosphate) to adjust for an equal P input of (47 kg ha^{-1}) in all treatments. There was no risk for K deficiency since minimal 100 kg K_2O ha^{-1} was added by the products. The pots (h = 15 cm, r = 11.3) were filled with soil to reach a bulk density of 1.4 g cm^{-3}. The pots were filled in two steps: half of the soil was added directly to the pot, after which the other half was filled with a mixture of soil, product and/or fertilizer. The pots were perforated at the bottom and placed on trays, allowing to add water in the trays during the experiment and to prevent nutrient leaching. On day seven, 50 seeds per pot were sown. The above ground plant material was harvested on day 35 (first cut), day 61 (second cut), day 90 (third cut) and day 120 (fourth cut). The DM yield of the separate cuts was measured by drying the plant material 48 h in a ventilated oven at 70°C. Afterwards, the total plant biomass from the four cuts per treatment and replicate were combined and ground (< 1 mm) in a plant mill. The total N concentration of the grass was determined according to EN 13654-2, with a Thermo scientific-flash 4000 total N analyzer, following the Dumas method. To calculate the NFRV (i.e., the mineral N fertilization rate corresponding with the calculated plant N uptake of the treatment) of the products, a crop

response curve was plotted as a linear regression of the total plant N uptake (N concentration * DM crop yield of the grass) in function of the mineral N fertilization rates in the control soil. Furthermore, at the end of the experiment, soil samples were taken in each pot to determine mineral N and pH-KCl (methods in section 6.2.2).

6.2.7. Statistical analysis

A one-way ANOVA and post-hoc Scheffé test were used ($p = 0.05$) to assess the significant differences in product quality between the treatments. A two-way ANOVA with factors 'treatment' and 'time' at $p = 0.05$ was used to test the evolution of L-TAS, H-TAS and X-TAS spores with composting. If there was a significant interaction between the effects of treatment and time on the spores, Tukey's HSD post hoc tests were carried out. A linear regression was conducted on the data from the N incubation experiment and the stability parameters to check if the data could be explained by a linear model ($p < 0.05$). An independent samples *t*-test was used to compare the silage treatments on field and lab scale (SF+S+G_E and SF+S+G_E_{lab}). Data analysis was carried out using IBM SPSS Statistics for Windows, version 21.0 (IBM Corp., Armonk, NY, USA).

6.3. Results and Discussion

6.3.1. Indicators for product stability of composts and silages

The evolution of different stability parameters was compared. Composting SF, both with or without clinoptilolite or bulking agents, resulted in an increase in pile temperature compared to the start (Figure 6.1A), NO_3^--N content, NO_3^--N/NH_4^+-N ratio and relative lignin content; and a decrease in OM and NH_4^+-N content (except for SF+CFM_C), C/N ratio, OUR, relative amount of (hemi-)cellulose and biodegradation potential (Table 6.1), all pointing towards OM degradation and stabilization. The relative amount of hemicellulose mainly decreased during the first month (most easily degradable component), while the relative amount of cellulose and lignin respectively decreased and increased, especially during the second month (Figure 6.1), as observed for other composts (Lashermes et al., 2012). The C/N ratios were below 15 and the OUR of all composts were pointing towards 'very stable composts' according to Belgian standards (value < 5 mmol kg^{-1} OM h^{-1}) (Table 6.1). Nevertheless, the composts could not be considered as stable after two months composting because they (1) still had temperatures above ambient air temperature (Figure 6.1A), which

was in line with the studies of Brito et al. (2008; 2012) where ambient air temperatures were reached only after 3-5 months composting.

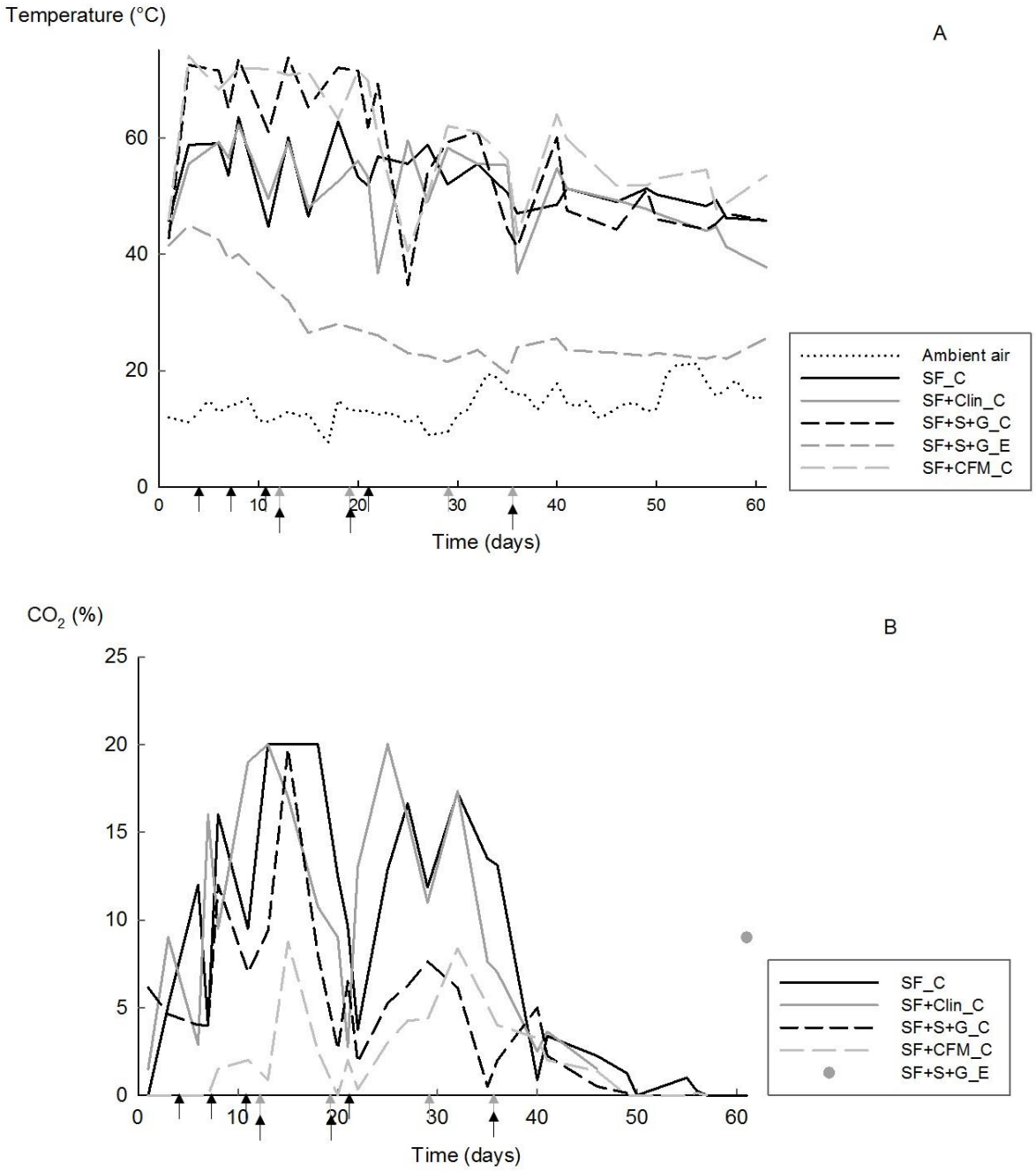


Figure 6.1 Temperature profiles (A) and CO_2 concentrations (B) of the composted solid fraction (SF) (SF_C), composted SF with clinoptilolite (SF+Clin_C), co-composted SF with cattle farmyard manure (SF+CFM_C), co-composted SF with straw and grass (SF+S+G_C) and co-ensiled SF with straw and grass (SF+S+G_E) (mean of 4 measurements, standard deviations were below 10°C and 5% CO_2). Arrows are indicating when piles were turned: on day 5, 7, 11, 13, 19, 21 and 36 for SF+S+G_C and SF+CFM_C (black arrows) and on day 13, 19, 29 and 36 for SF_C and SF+Clin_C (grey arrows).

Table 6.1 Physico-chemical quality at day 0, 32 and 61 of the experiment (mean ± standard deviation, $n = 4$) for composted solid fraction (SF) (SF_C), composted SF with clinoptilolite (SF+Clin_C), co-composted SF with cattle farmyard manure (SF+CFM_C), co-composted SF with straw and grass (SF+S+G_C) and co-ensiled SF with straw and grass (SF+S+G_E). Treatments indicated with the same letter in the same sampling time, are not significantly different (Scheffé test, $p < 0.05$). DM = dry matter, OM = organic matter. The silage treatment was not sampled at Day 32 in order not to disturb the treatment. No weeds germinated after 21 days incubation, regardless of the treatment.

	Day 0		Day 32		Day 61		Day 0		Day 32		Day 61		Day 0		Day 32		Day 61	
	pH-H_2O (-)						Electrical conductivity (µS cm^{-1})						Fresh bulk density (kg m^{-3})					
SF_C	7.7 ± 0.1	b	9.0 ± 0.1	a	8.8 ± 0.1	b	2113 ± 116	a	1846 ± 106	a	1864 ± 252	b	355 ± 6	c	468 ± 5	b	514 ± 6	d
SF+Clin_C	7.7 ± 0.1	b	9.0 ± 0.1	a	8.7 ± 0.3	ab	2113 ± 116	a	1792 ± 73	a	1696 ± 52	b	355 ± 6	c	474 ± 5	b	518 ± 6	d
SF+CFM_C	8.3 ± 0.0	c	9.3 ± 0.0	c	8.8 ± 0.0	ab	2325 ± 125	ab	1904 ± 45	a	3030 ± 124	c	246 ± 11	a	340 ± 22	a	350 ± 16	b
SF+S+G_C	6.6 ± 0.1	a	9.1 ± 0.0	b	8.7 ± 0.1	ab	2595 ± 206	b	2430 ± 72	b	3260 ± 181	c	295 ± 4	b	361 ± 16	a	417 ± 9	c
SF+S+G_E	6.6 ± 0.1	a			8.5 ± 0.1	a	2595 ± 206	b			1243 ± 14	a	295 ± 4	b			199 ± 27	a
	Dry matter content (% of fresh weight)						Organic matter content (% of DM)						Oxygen uptake rate (mmol kg^{-1} OM h^{-1})					
SF_C	24.0 ± 0.3	a	24.0 ± 0.3	a	26.1 ± 0.3	b	78.9 ± 2.1	a	69.5 ± 1.4	b	62.8 ± 1.0	ab	36.4 ± 28.3	a	2.8 ± 0.3	a	1.7 ± 0.6	a
SF+Clin_C	24.0 ± 0.3	a	23.0 ± 1.3	a	25.5 ± 1.0	b	78.9 ± 2.1	a	66.1 ± 0.9	a	60.8 ± 1.0	a	36.4 ± 28.3	a	4.4 ± 0.7	a	3.4 ± 0.2	a
SF+CFM_C	27.3 ± 0.1	c	36.5 ± 1.6	c	47.1 ± 1.2	d	80.3 ± 0.5	a	69.7 ± 0.7	b	64.1 ± 1.2	b	9.2 ± 6.6	a	4.0 ± 0.6	a	4.6 ± 0.5	a
SF+S+G_C	25.6 ± 0.3	b	32.6 ± 1.4	b	35.4 ± 3.8	c	84.7 ± 1.5	b	70.0 ± 1.2	b	62.3 ± 1.0	ab	31.4 ± 22.7	a	5.4 ± 2.1	a	2.8 ± 0.1	a
SF+S+G_E	25.6 ± 0.3	b			20.4 ± 1.2	a	84.7 ± 1.5	b			83.7 ± 1.9	c	31.4 ± 22.7	a			11.2 ± 2.1	b
	NO_3^--N (mg kg^{-1} DM)						NH_4^+-N (mg kg^{-1} DM)						NO_3^--N / NH_4^+-N (-)					
SF_C	14 ± 6	b	11 ± 13	a	1023 ± 97	d	2166 ± 246	b	4158 ± 59	c	1232 ± 117	a	0.007 ± 0.002	a	0.003 ± 0.003	a	0.835 ± 0.112	c
SF+Clin_C	14 ± 6	b	4 ± 3	a	1259 ± 307	d	2166 ± 246	b	4422 ± 454	c	924 ± 369	a	0.007 ± 0.002	a	0.001 ± 0.001	a	1.660 ± 1.002	c
SF+CFM_C	6 ± 1	a	8 ± 13	a	56 ± 15	b	567 ± 39	a	338 ± 232	a	562 ± 35	a	0.011 ± 0.001	a	0.048 ± 0.084	a	0.100 ± 0.031	ab
SF+S+G_C	20 ± 10	b	3 ± 1	a	254 ± 66	c	1738 ± 396	b	1332 ± 447	b	1137 ± 269	a	0.013 ± 0.009	a	0.002 ± 0.001	a	0.238 ± 0.099	b
SF+S+G_E	20 ± 10	b			5 ± 2	a	1738 ± 396	b			5589 ± 790	b	0.013 ± 0.009	a			0.001 ± 0.000	a
	Total N (mg kg^{-1} DM)						Total P (mg kg^{-1} DM)						C/N (-)					
SF_C	21805 ± 776	b	24468 ± 970	a	30845 ± 1436	a	7064 ± 482	b	9042 ± 131	b	11625 ± 258	c	20.1 ± 0.8	b	15.8 ± 0.9	b	11.3 ± 0.3	a
SF+Clin_C	21805 ± 776	b	23765 ± 1005	a	28730 ± 2415	a	7064 ± 482	b	8701 ± 697	ab	10689 ± 508	b	20.1 ± 0.8	b	15.5 ± 0.8	b	11.8 ± 1.0	a
SF+CFM_C	19585 ± 222	a	28040 ± 1170	b	32450 ± 759	a	5329 ± 152	a	8022 ± 339	a	9818 ± 176	b	22.8 ± 0.1	c	13.8 ± 0.6	a	11.0 ± 0.8	a
SF+S+G_C	25765 ± 1242	c	29100 ± 982	b	33190 ± 1844	a	5336 ± 291	a	9594 ± 226	b	11729 ± 231	c	18.3 ± 0.6	a	13.4 ± 0.5	a	10.5 ± 0.8	a
SF+S+G_E	25765 ± 1242	c			29950 ± 3584	a	5336 ± 291	a			5112 ± 475	a	18.3 ± 0.6	a			15.7 ± 1.6	b
	C/P (-)						N/P (-)						Hemicellulose (% of OM)					
SF_C	62.4 ± 5.9	a	42.7 ± 1.3	a	30.0 ± 0.9	c	3.1 ± 0.2	a	2.7 ± 0.1	a	2.7 ± 0.1	a	24.0 ± 4.7	a	14.3 ± 5.2	a	11.6 ± 7.4	a
SF+Clin_C	62.4 ± 5.9	a	42.3 ± 2.9	a	31.6 ± 1.7	c	3.1 ± 0.2	a	2.7 ± 0.3	a	2.7 ± 0.3	a	24.0 ± 4.7	a	19.8 ± 4.5	b	15.4 ± 2.1	a
SF+CFM_C	83.8 ± 3.0	b	48.4 ± 2.0	b	36.3 ± 2.8	b	3.7 ± 0.1	a	3.5 ± 0.2	a	3.3 ± 0.1	a	23.3 ± 1.5	a	16.6 ± 2.4	a	11.8 ± 2.0	a
SF+S+G_C	88.4 ± 6.4	c	40.5 ± 0.5	a	29.5 ± 1.0	c	4.8 ± 0.5	b	3.0 ± 0.1	a	2.8 ± 0.1	a	24.2 ± 5.2	a	16.3 ± 2.2	a	14.8 ± 6.0	a
SF+S+G_E	88.4 ± 6.4	c			91.7 ± 10.8	a	4.8 ± 0.5	b			5.9 ± 1.2	b	24.2 ± 5.2	a			26.2 ± 2.0	b
	Cellulose (% of OM)						Lignin (% of OM)						Biodegradation potential (-)					
SF_C	38.4 ± 2.9	b	34.1 ± 3.6	b	21.0 ± 5.4	a	16.7 ± 0.5	c	26.1 ± 1.3	a	43.7 ± 2.3	c	3.7 ± 0.2	a	1.9 ± 0.3	b	0.7 ± 0.2	a
SF+Clin_C	38.4 ± 2.9	b	32.5 ± 3.3	b	20.5 ± 3.4	a	16.7 ± 0.5	c	26.3 ± 1.0	a	44.0 ± 3.0	c	3.7 ± 0.2	a	2.0 ± 0.3	c	0.8 ± 0.1	a
SF+CFM_C	39.5 ± 1.4	b	26.9 ± 1.6	a	13.3 ± 2.3	a	14.6 ± 0.8	b	25.1 ± 1.2	a	37.4 ± 2.3	b	4.3 ± 0.3	a	1.7 ± 0.2	a	0.7 ± 0.1	a
SF+S+G_C	31.6 ± 3.7	a	34.0 ± 2.0	b	13.1 ± 4.6	a	9.6 ± 1.4	a	26.4 ± 0.5	a	39.9 ± 1.2	bc	5.9 ± 0.9	b	1.9 ± 0.2	bc	0.7 ± 0.2	a
SF+S+G_E	31.6 ± 3.7	a			33.6 ± 3.0	b	9.6 ± 1.4	a			17.7 ± 2.5	a	5.9 ± 0.9	b			3.8 ± 0.5	b

Furthermore, (2) the NO_3^--N/NH_4^+-N ratios were < 1 (except for SF+Clin_C) and (3) the NH_4^+-N contents were > 400 mg kg^{-1} DM for all composts (Table 6.1) indicating oxygen shortage, as was confirmed by the high CO_2 measurements (Figure 6.1B). Considering the silages, the end product of the co-ensiled SF at field scale (Table 6.1) and the silages from the lab experiment (Table 6.2) could be considered stable based on the C/N ratios < 20 and OUR < 15, while the NH_4^+-N concentrations were very high, the NO_3^--N/NH_4^+-N ratios were < 1 and the biodegradation potential between 1.6 and 3.9, rather indicating a low stability. Previous research by Hutchinson and Griffin (2008) showed that the C/N ratio of the end product can be an inappropriate indicator of stability because it is mainly affected by the initial C/N ratio of the feedstock (Nolan et al., 2011) and it can level off before the compost has stabilized (Zmora-Nahum et al., 2005). However, it was remarkable that the OUR, a general accepted stability indicator for composts, categorized the composts and especially the silages as stable. In contrast, in the study of Viaene et al. (2016a) ensiling cattle farmyard manure resulted in an unstable product with an OUR between 24-30 mmol kg^{-1} OM h^{-1} and a biodegradation potential of 4.1. Generally, measurements of microbial respiration (such as OUR) can be very sensitive to changes of moisture, temperature, oxygen and N availability (Veeken et al., 2007). However, moisture, temperature and oxygen availability during the OUR test in our study were in line with the protocol for OUR analysis and the products were high in N content. It is possible that despite the used pH-buffer (pH of 6.5) in the OxiTop test, the high pH of the products (8.5-8.8) inhibited the initial (mesophilic) aerobic activity to start up the aerobic degradation, resulting in low microbial activity and thus low oxygen uptake rates and 'apparent' stable products. This is illustrated by plotting the pH of the products against the OUR values (Figure 6.2). It indeed showed that the product pH strongly affected the OUR, i.e., a pH higher than 8.2 resulted in a maximum OUR of 24 mmol kg^{-1} OM h^{-1}.

Table 6.2 Physico-chemical quality of the end products (mean ± standard deviation, $n = 4$) for ensiled products at lab scale: solid fraction (SF) (SF_E_{lab}), ensiled SF with clinoptilolite (SF+Clin_E_{lab}), co-ensiled SF with cattle farmyard manure (SF+CFM_E_{lab}) and co-ensiled SF with straw and grass (SF+S+G_E_{lab}). Treatments indicated with the same letter are not significantly different (Scheffé test, $p < 0.05$). DM = dry matter, OM = organic matter.

	SF_E_{lab}		SF+Clin_E_{lab}		SF+CFM_E_{lab}		SF+S+G_E_{lab}	
pH-H_2O (-)	9.1 ± 0.1	a	9.0 ± 0.1	a	9.2 ± 0.1	a	7.3 ± 0.8	a
pH-KCl (-)	8.8 ± 0.3	b	8.9 ± 0.1	b	9.2 ± 0.1	b	7.8 ± 0.7	a
Electrical conductivity (µS cm^{-1})	1451 ± 53	b	1335 ± 48	b	1007 ± 51	a	1411 ± 80	b
Fresh bulk density (kg m^{-3})	373 ± 9	b	375 ± 12	b	187 ± 4	a	185 ± 10	a
Dry matter content (% of fresh weight)	24.1 ± 1.4	ab	25.2 ± 0.4	bc	27.0 ± 1.0	c	22.3 ± 1.4	a
Organic matter content (% of DM)	77.6 ± 1.3	ab	75.5 ± 1.5	a	80.0 ± 1.6	bc	82.3 ± 0.9	c
Oxygen uptake rate (mmol kg^{-1} OM h^{-1})	6.5 ± 1.3	a	7.0 ± 0.4	a	7.2 ± 0.1	a	38.7 ± 16.4	a
NO_3^--N (mg kg^{-1} DM)	7 ± 2	a	6 ± 2	a	7 ± 5	a	2 ± 1	a
NH_4^+-N (mg kg^{-1} DM)	3057 ± 869	a	2976 ± 1457	a	1563 ± 397	a	8825 ± 2129	b
Total N (mg kg^{-1} DM)	22538 ± 2465	a	21525 ± 1555	a	22845 ± 1820	a	25690 ± 1679	a
Total P (mg kg^{-1} DM)	6822 ± 248	b	7224 ± 587	b	5497 ± 546	a	5770 ± 229	a
C/N (-)	19.3 ± 2.5	a	19.6 ± 1.5	a	19.6 ± 1.7	a	17.9 ± 1.2	a
C/P (-)	63.3 ± 2	a	58.4 ± 5.7	a	81.6 ± 10.3	b	79.3 ± 3.9	b
N/P (-)	3.3 ± 0.4	a	3.0 ± 0.2	a	4.2 ± 0.3	b	4.5 ± 0.3	b
Total K (mg kg^{-1} DM)	22307 ± 695	ab	21366 ± 1427	a	25741 ± 1822	bc	27059 ± 2154	c
Total Mg (mg kg^{-1} DM)	6241 ± 162	b	6391 ± 745	b	4929 ± 515	a	5153 ± 28	a
Total Ca (mg kg^{-1} DM)	16194 ± 466	b	17408 ± 1485	b	12414 ± 1552	a	11960 ± 493	a
Total Na (mg kg^{-1} DM)	1558 ± 39	bc	1364 ± 140	ab	1259 ± 46	a	1637 ± 119	c
Hemicellulose (% of OM)	21.3 ± 4.5	a	22.2 ± 2.7	a	20.7 ± 2.0	a	22.7 ± 3.4	a
Cellulose (% of OM)	27.6 ± 2.4	ab	25.0 ± 3.1	a	29.6 ± 2.3	b	35.4 ± 1.2	c
Lignin (% of OM)	28.0 ± 1.1	c	29.1 ± 2.6	c	22.3 ± 2.3	b	15.0 ± 0.4	a
Potential biodegradability (-)	1.8 ± 0.2	a	1.6 ± 0.2	a	2.3 ± 0.3	b	3.9 ± 0.3	c

All stability indicators indicate a clear distinction between composts and silages: silages had a much higher biodegradation potential (> 1.6), OUR (> 6.5 mmol kg^{-1} OM h^{-1}), C/N ratio (> 15.7), NH_4^+-N concentration (> 1563 mg kg^{-1} DM) and lower NO_3^--N/NH_4^+-N ratio (< 0.005) compared to the composts, and thus could be considered as less stable. Furthermore, linear regression analyses between the conventional stability indicators and the biodegradation potential of the composts and silages showed different results for silages and composts. For silages, we found a positive linear relationship between biodegradation potential and OUR ($p = 0.03$, $R^2 = 0.46$) and NH_4^+-N content ($p < 0.01$, $R^2 = 0.60$) and a negative relationship with the C/N ratio ($p = 0.02$, $R^2 = 0.28$). For composts, there was only a positive relationship with the NO_3^--N/NH_4^+-N ratio ($p = 0.03$, $R^2 = 0.31$). Hence, both ensiling and composting are biological processes of microbial decomposition, but the dynamics for N transformation are different. Composting mineralizes the organic N into NH_4^+-N and then NO_3^--N as aerobic conditions prevail, while ensiling results in higher NH_4^+-N concentrations but nitrification does not occur due to the anaerobic conditions.

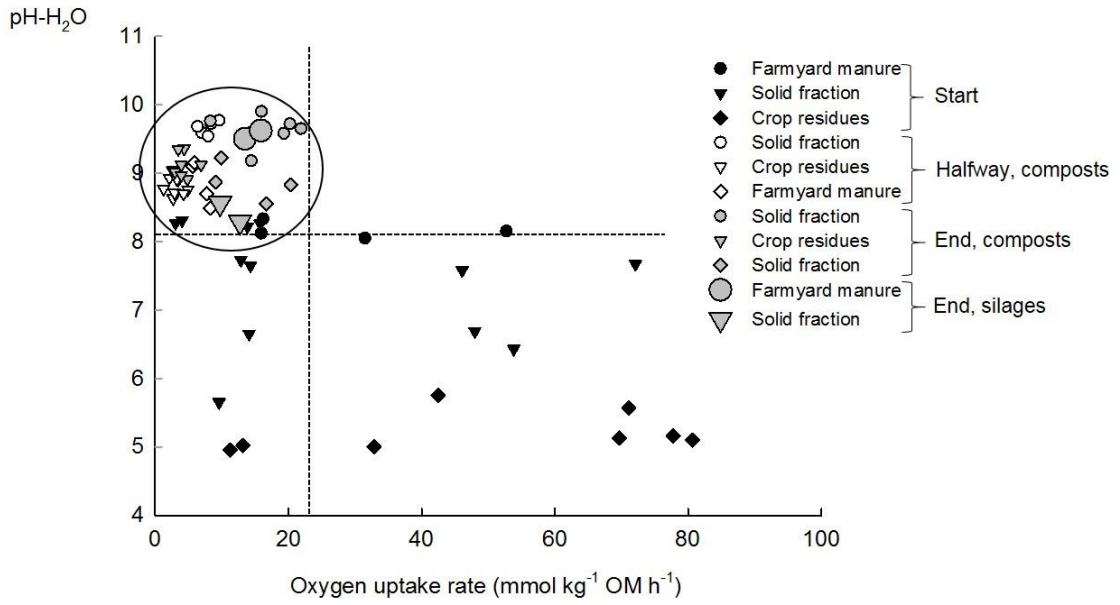


Figure 6.2 Relation between the oxygen uptake rate (OUR) and pH-H_2O of the feedstock mixtures at the start, of the composts halfway and of the composts and silages at the end of the experiment (day 61). Data from a simultaneous compost experiment with leek residues and cattle farmyard manure were also included. pH-values > 8.2 resulted in OUR < 24 mmol kg^{-1} OM h^{-1}.

6.3.2. Ensiling experiments

In the field scale experiment, the temperature of SF+S+G_E dropped during the first 20 days from 45°C until approximately 23°C, after which it remained constant (Figure 6.1A). Moreover, the average pile temperature of SF+S+G_E was approximately half that of SF+S+G_C, explained by the high CO_2 concentrations, indicative for shortage in oxygen (Figure 6.1B) and hence a lower aerobic decomposition of OM, as confirmed by the lower fresh weight losses (5% vs. 61% for respectively SF+S+G_E and SF+S+G_C, Table 6.3) and smaller decrease in OUR and biodegradation potential compared to SF+S+G_C (Table 6.1). However, the considerable losses of dry weight, OM, N (Table 6.3) and the decreased OUR and biodegradation potential (Table 6.1) indicated that also with co-ensiling OM degradation took place, although limited. The end product of SF+S+G_E had a lower DM content and higher OM content (DM = 20.4% of fresh weight, OM = 83.7% of DM) compared to SF+S+G_C (DM = 35.4% of fresh weight, OM = 62.3% of DM) (Table 6.1). Similar to the study of Thomsen (2000a) and Viaene et al. (2016a), the composted and ensiled manure ended up with a same total N content, but the ensiled manure had a significantly higher proportion of mineral N. (Co-)ensiling did not result in a decrease in pH (Table 6.1), which normally occurs during fermentation of dairy manure (Aguerre et al.,

2012). Too high NH_4^+-N concentrations or a shortage of fermentable sugars may have limited the fermentation process.

Table 6.3 Initial and end fresh weight (FW), dry weight (DW) and organic matter (OM) content (in kg) and the corresponding relative loss for composted solid fraction (SF) (SF_C), composted SF with clinoptilolite (SF+Clin_C), co-composted SF with cattle farmyard manure (SF+CFM_C), co-composted SF with straw and grass (SF+S+G_C) and co-ensiled SF with straw and grass (SF+S+G_E).

	Start (kg)			End (kg)			Loss (%)		
	FW	DW	OM	FW	DW	OM	FW	DW	OM
SF_C	13420	3214	2536	7368	1921	1206	45	40	52
SF+Clin_C	14280	3420	2698	7952	2024	1230	44	41	54
SF+CFM_C	11180	3055	2453	4427	2085	1336	60	32	46
SF+S+G_C	10434	2671	2261	4115	1455	906	61	46	60
SF+S+G_E	10176	2605	2205	9672	1976	1654	5	24	25

As discussed in section 6.3.1, co-ensiling did not result in a stable end product, but conserved C and nutrients. The product is expected to further decompose once the silage is opened and applied on the soil. Compared to the initial SF, ensiling SF with straw and grass increased the NH_4^+-N concentration, C/P and N/P ratios and decreased the total P content and NO_3^--N/NH_4^+-N and C/N ratio. As such, through silage application, more N and C can be added to the soil per unit of P. Moreover, the end product of SF+S+G_E had a higher C/N (15.7), C/P (91.7) and N/P (5.9) ratio compared to a C/N = 10.5, C/P = 29.5 and N/P = 2.8 for SF+S+G_C (Table 6.1). Therefore, silages are relatively more suited for local use, while the SF compost is more suited for export, as more P is concentrated per tonne of fresh matter.

Product parameters were similar for co-ensiling SF at field (SF+S+G_E, Table 6.1) and lab (SF+S+G_E_{lab}) scale (Table 6.4). However, SF+S+G_E_{lab} had a much higher NH_4^+-N concentration compared to SF+S+G_E. This could be due to better ensiling conditions at lab scale (more airtight). Consequently, the lab scale experiments with ensiling buckets can be considered as a reliable and efficient technique to perform ensiling experiments. Similarly to ensiling at field scale, the temperature at the end of the silage experiments on lab scale was on average 18°C. Fresh weight losses on lab scale for the silage treatments were even smaller (< 2%) compared to ensiling at field scale (5%, Table 6.3), and there was no differences among treatments. Furthermore, there were no sap losses, however, in the field scale experiment we noticed some sap losses when opening the silage. An overview of the end product characterization of the different silages from the lab experiment can be found in Table 6.2. In general, compared to composting (Table 6.1), ensiling resulted in a product with a lower EC, DM content, NO_3^--N concentration, total nutrient concentrations (N, P, K,

Mg, Ca, Na) and a higher OM content, NH_4^+-N concentration, C/N and C/P ratio, OUR and biodegradation potential, i.e., a lower product stability.

Table 6.4 Comparison between end product quality of the co-ensiled solid fraction (SF) with straw and grass at field (SF+S+G_E) and lab scale (SF+S+G_E_{lab}). Parameters indicated with an asterisk are significantly different between the experiments (independent samples *t*-test, $p < 0.05$). DM = dry matter, OM = organic matter.

	SF+S+G_E	SF+S+G_E_{lab}	
pH-H_2O (-)	8.5 ± 0.1	7.3 ± 0.8	
Electrical conductivity (µS cm^{-1})	1243 ± 14	1411 ± 80	*
Fresh bulk density (kg m^{-3})	199 ± 27	185 ± 10	
Dry matter content (% of fresh weight)	20.4 ± 1.2	22.3 ± 1.4	
Organic matter content (% of DM)	83.7 ± 1.9	82.3 ± 0.9	
Oxygen uptake rate (mmol kg^{-1} OM h^{-1})	11.2 ± 2.1	38.7 ± 16.4	
NO_3^--N (mg kg^{-1} DM)	5 ± 2	2 ± 1	
NH_4^+-N (mg kg^{-1} DM)	5589 ± 790	8825 ± 2129	*
Total N (mg kg^{-1} DM)	29950 ± 3584	25690 ± 1679	
Total P (mg kg^{-1} DM)	5112 ± 475	5770 ± 229	
C/N (-)	15.7 ± 1.6	17.9 ± 1.2	
C/P (-)	91.7 ± 10.8	79.3 ± 3.9	
N/P (-)	5.9 ± 1.2	4.5 ± 0.3	
Hemicellulose (% of OM)	26.2 ± 2.0	22.7 ± 3.4	
Cellulose (% of OM)	33.6 ± 3.0	35.4 ± 1.2	
Lignin (% of OM)	17.7 ± 2.5	15.0 ± 0.4	
Potential biodegradability (-)	3.4 ± 0.5	3.9 ± 0.3	

6.3.3. Composting experiments

The SF was typically characterized by a high moisture content (76% on fresh weight), low C/N ratio (20.1) and limited pore space in comparison to other feedstock mixtures for composting. High CO_2 concentrations (> 16%), were measured during composting pure SF (Figure 6.1B), indicating a limited oxygen availability. Turning on day 13, 19, 29 and 36 could not control these excessively high CO_2 levels in the piles (SF_C and SF+Clin_C). A frequent need for turning can be considered as time-consuming, fuel-consuming and cost-inefficient, and hence might be a barrier to on-farm composting (Viaene et al., 2016b). Due to aeration, temperatures up to 65°C were reached (below export requirements) in both piles (Figure 6.1A), comparable to the study of Brito et al. (2008). Adding straw/grass (SF+S+G_C) or CFM **(**SF+CFM_C) to the SF (SF_C) led to a drier feedstock mixture at the start (Table 6.1), which could have facilitated oxygen transport (Brito et al., 2012). This was confirmed by the lower CO_2 levels in the piles (Figure 6.1B). SF+CFM_C and SF+S+G_C had average CO_2 levels of 2 and 5% respectively, while SF_C had an average of 8-9% CO_2

over the whole period. Starting from day 36 of the composting process, the CO_2 levels were below 8% for all treatments. Temperatures in the co-composted SF treatments were significantly higher (maximum 74°C, higher than the required 70°C for export) during the first 20 days, compared to composting pure SF (maximum 65°C), probably as a consequence of the higher oxygen availability and hence higher microbial activity in the co-composted piles (Caceres et al., 2006) (Figure 6.1A). This was in line with co-composting SF of pig slurry (Nolan et al., 2011), but in contrast to other studies where lower temperatures were noticed with co-composting SF of cattle slurry digestate with vine shoot prunings (Bustamante et al., 2012) and co-composting SF of dairy slurry with straw (Michel et al., 2004), compared to composting without bulking agents. The lower temperatures in those studies were due to the increased free air space, followed by a greater convective air flow and heat loss (Michel et al., 2004) and the high lignin content of the bulking agent, slowing down the degradation of the mixture by the microorganisms (Bustamante et al., 2012). However, in our case, also N-rich byproducts (fresh grass clippings or manure) were added with the straw, resulting in a higher N content, a higher proportion of easily biodegradable OM and an increased biodegradation potential (Table 6.1) of the feedstock mixture at the start (increased percentage hemicellulose and decreased percentage cellulose and lignin), leading to higher pile temperatures. Pile temperatures rose above 55°C for 15 days for SF+S+G_C and SF+CFM_C (Figure 6.1A), potentially indicating the destruction of pathogens (Lung et al., 2001) and weed seeds (Eghball and Lesoing, 2000), important as a requirement for export. Indeed, no weeds germinated after amending the composts to peat (Table 6.1), however, destruction of pathogens was not validated in this experiment. We did evaluate L-TAS, H-TAS and X-TAS spores before and after composting. X-TAS spores were below or close to the detection limit in all feedstock materials prior to composting (Table 6.5). Composting increased L-TAS and H-TAS spores in SF_C, SF+S+G_C and CFM_C by 0.5, 1.4 and 0.4 $\log_{10}$ units, respectively. X-TAS spores were not detected in SF_C and SF+S+G_C, in contrast to composted CFM which contained almost 5 $\log_{10}$ cfu g^{-1} of X-TAS spores. These results indicate growth and sporulation of X-TAS bacteria during composting of CFM, but not during (co-)composting SF in the trials described here. Additional investigations have shown high concentrations of X-TAS spores in a minority of batches of stored SF at dairy farms, showing signs of spontaneous composting (Driehuis et al., 2015). These findings indicate that SF can be a suitable substrate for growth and sporulation of X-TAS bacteria under certain conditions. Factors that control growth and

sporulation of X-TAS bacteria during composting of SF and CFM are not well understood yet.

Table 6.5 Concentrations of low-heat-resistant spores of thermophilic aerobic spore-forming bacteria (L-TAS), high-heat-resistant spores of thermophilic aerobic spore-forming bacteria (H-TAS) and extreme-heat-resistant spores of thermophilic aerobic spore-forming bacteria (X-TAS) prior to composting (day 0) and after 61 days (mean ± standard deviation, $n = 3$) for composted solid fraction (SF_C), co-composted SF with straw and grass (SF+S+G_C) and composted cattle farmyard manure (CFM_C). There was an increase in L-TAS and H-TAS spores after composting compared to the initial feedstock mixture in all treatments. No X-TAS spores were detected after (co-)composting SF (two-way ANOVA, Tukey's HSD post hoc test, $p < 0.05$).

	L-TAS (log_{10} CFU g^{-1})			H-TAS (log_{10} CFU g^{-1})			X-TAS (log_{10} CFU g^{-1})		
	Day 0	Day 61	*p*	Day 0	Day 61	*p*	Day 0	Day 61	*p*
SF_C	6.8 ± 0.1	7.3 ± 0.1	0.01	5.9 ± 0.1	6.4 ± 0.2	0.01	< 1.0	< 2.0	
SF+S+G_C	6.7 ± 0.1	8.1 ± 0.2	< 0.01	6.0 ± 0.1	7.4 ± 0.1	< 0.01	1.2 ± 0.2	< 2.0	
CFM_C	7.4 ± 0.2	7.8 ± 0.1	0.03	7.2 ± 0.1	7.6 ± 0.1	< 0.01	1.2 ± 0.4	4.8 ± 0.6	< 0.01

During the experiment the DM content increased more strongly for SF+CFM_C (a relative increase of 73%) and for SF+S+G_C (+38%) compared to SF_C (+9%) (Table 6.1). The higher temperatures, evaporation and turning frequency (day 5, 7, 11, 13, 19, 21 and 36, based on temperatures > 65°C) are explaining the higher fresh weight losses (Table 6.3) and drier end products of SF+S+G_C and SF+CFM_C (Table 6.1). Compared to SF_C, SF+S+G_C and SF+CFM_C resulted in a lower NH_4^+-N content after 32 days, while the total N content had increased. After 61 days, SF+S+G_C and SF+CFM_C resulted in less NO_3^--N and a lower NO_3^--N/NH_4^+-N ratio, but an equal total N content. The end product of SF+CFM_C (9.8 kg P $tonne^{-1}$ DM) had a lower P content and higher C/P ratio than SF_C (11.6 kg P $tonne^{-1}$ DM) (Table 6.1). Consequently, co-composting with CFM increased the value for local use of the SF, because more C per unit of P can be added to the soil. Contrary to our hypothesis, adding clinoptilolite to the SF did not have an effect on the mineral N content of the compost, but there was a trend towards a higher NO_3^--N/NH_4^+-N ratio, even higher than 1 (Table 6.1), with an equal total mineral N content, indicating a shift from NH_4^+-N to NO_3^--N (nitrification was not due to temperatures < 40°C, Figure 6.1A) and thus a more reactive composting process. This might have decreased the risk for gaseous NH_3 losses.

Compared with the initial SF, composting with or without bulking agent resulted in a decreased OM content, NH_4^+-N concentration, OUR, biodegradation potential, C/N, C/P and N/P ratios and an increased NO_3^--N concentration, NO_3^--N/NH_4^+-N ratio and total P content (Table 6.1). Therefore, composting SF is more suited for export, because P is more concentrated (higher P content per tonne fresh matter) and the product is more stabilized and sanitized compared to ensiled SF.

6.3.4. N mineralization and N fertilizer replacement value

During the N mineralization experiment soil NH_4^+-N was negligible for all treatments at each sampling time except immediately after incorporation of the products. The mineral N in the control soil showed a typical linear increase during the incubation period (data not shown). Soil pH-KCl decreased immediately after incorporation of the products, but thereafter remained constant around 6 for SF_C (6.04 ± 0.03), SF+Clin_C (6.04 ± 0.04) and SF+S+G_C (6.02 ± 0.02).

During the NFRV experiment, adding N fertilizer to the control soil resulted in an acidifying effect, however, there was no difference in soil pH-KCl at the end of the experiment between the different treatments (Table 6.6). Furthermore, at the end of the experiment, no effect of treatment on the soil mineral N content was noticed (Table 6.6). A clear plant response on the mineral N doses was observed (Table 6.6). The low soil mineral N concentrations at the end are indicating that most of the supplied N was used by the grass. This means N was the limiting factor for crop growth, consequently, a clear N effect was measured in the pot experiment. Based on mineral N fertilizer input and N uptake of treatments CON, CON_{50N}, CON_{100N} and CON_{150N}, a N response curve was calculated: $N_{uptake} = 33.11 + 0.62 * N_{mineral\ fertilizer}$. With N_{uptake} and $N_{mineral\ fertilizer}$ in kg ha^{-1} ($R^2 = 0.97$, $p < 0.01$).

Table 6.6 Dry matter crop yield, plant N uptake and soil characteristics (mineral N content and pH-KCl) at the end of the perennial rye grass pot trial (mean ± standard deviation, $n = 3$) for composted solid fraction (SF) (SF_C), composted SF with clinoptilolite (SF+Clin_C), co-composted SF with straw and grass (SF+S+G_C) and co-ensiled SF with straw and grass (SF+S+G_E). Treatments indicated with the same letter are not significantly different (Scheffé test, $p < 0.05$), for NFRV control treatments were excluded from the statistical analysis.

	Dry matter crop yield		N uptake		NFRV		Mineral N		pH-KCl	
	kg ha^{-1}		kg ha^{-1}		kg kg^{-1} N		kg ha^{-1}		-	
CON	2107 ± 189	a	31 ± 3		-		2.4 ± 0.5	a	6.11 ± 0.09	c
CON_{50N}	3597 ± 157	c	67 ± 6		-		2.4 ± 0.3	a	6.07 ± 0.03	bc
CON_{100N}	4961 ± 66	e	97 ± 7		-		2.1 ± 0.4	a	5.89 ± 0.05	ab
CON_{150N}	5771 ± 106	f	125 ± 12		-		1.9 ± 0.1	a	5.71 ± 0.08	a
SF_C	2825 ± 86	b	50 ± 1	a	0.28 ± 0.02	a	1.8 ± 0.2	a	6.19 ± 0.06	c
SF+Clin_C	2928 ± 109	b	50 ± 2	a	0.26 ± 0.02	a	2.4 ± 0.7	a	6.25 ± 0.04	c
SF+S+G_C	2832 ± 96	b	55 ± 3	a	0.35 ± 0.04	a	3.1 ± 0.8	a	6.21 ± 0.06	c
SF+S+G_E	4242 ± 162	d	67 ± 2	b	0.55 ± 0.04	b	3.2 ± 0.8	a	6.08 ± 0.06	bc

In contrast to adding SF_C, adding SF+Clin_C did not result in significant N mineralization during the 100 days of incubation (Table 6.7), which could be related to the adsorption of NH_4^+-N by the clinoptilolite. However, there was no effect of clinoptilolite amendment on DM crop yield, plant N uptake and NFRV (Table 6.6). SF+S+G_C showed a lower N mineralization during the first 37 days compared to SF_C (Figure 6.3), related to the lower

NO_3^--N content of the compost (Table 6.1). However, this difference disappeared during the pot trial as there was no difference for N mineralization after 100 days (Figure 6.3), for crop growth, for plant N uptake and for NFRV after 120 days (Table 6.6) between the soils amended with composted SF or co-composted SF with structure-rich feedstock. Due to their low mineralization rates, composted as well as co-composted SF can be considered as slow-release fertilizers. Incorporation of SF+S+G_E resulted in a lower soil pH-KCl (5.87 ± 0.06) at the end of the incubation period compared to SF+S+G_C. Addition of extra N fertilizer (SF+S+G_E_{60N}) decreased pH-KCl (5.76 ± 0.01) compared to SF+S+G_E. During co-ensiling there was less loss of mineral N, resulting in more mineral N in the end product compared to co-composting (Table 6.1). Therefore, SF+S+G_E resulted in N mineralization twice as high (38 mg kg^{-1} or 37%) after 100 days compared to the same compost treatment (17 mg kg^{-1} or 17%) (Figure 6.3 and Table 6.7), despite the expected N immobilization in SF+S+G_E. Since extra N fertilization did not have an effect on the N mineralization (Figure 6.3), mineral N was sufficiently available from the silage and no N immobilization occurred as was the case with anaerobically stored slurry and straw (Sorensen, 1998), anaerobically stored straw-rich ruminant manure (Thomsen and Olesen, 2000) and silages of N-rich crop residues (Agneessens et al., 2015). The higher N mineralization was reflected by a higher DM crop yield, plant N uptake and NFRV with the silage compared to the compost treatment (NFRV of 0.55 and 0.35 kg kg^{-1} N, respectively for SF+S+G_E and SF+S+G_C) (Table 6.6).

Table 6.7 N mineralization (%) after 100 days incubation for the end products of composted solid fraction (SF) (SF_C), composted SF with clinoptilolite (SF+Clin_C), co-composted SF with straw and grass (SF+S+G_C), co-ensiled SF with straw and grass (SF+S+G_E) and co-ensiled SF with straw and grass + extra N fertilizer (SF+S+G_E_{60N}). The parameters a and b are respectively the intercept and slope from the linear regression of the relative N mineralization in function of time (N mineralization = a*time + b). $p > 0.05$ indicates no net N mineralization.

	a (% N day^{-1})	b (% N)	N mineralization after 100 days (%)	R^2	*p*
SF_C	0.06	13.23	19	0.67	0.045
SF+Clin_C	-	-	-	-	0.120
SF+S+G_C	0.10	8.19	17	0.83	0.012
SF+S+G_E	0.15	22.44	37	0.93	0.002
SF+S+G_E_{60N}	-	-	-	-	0.158

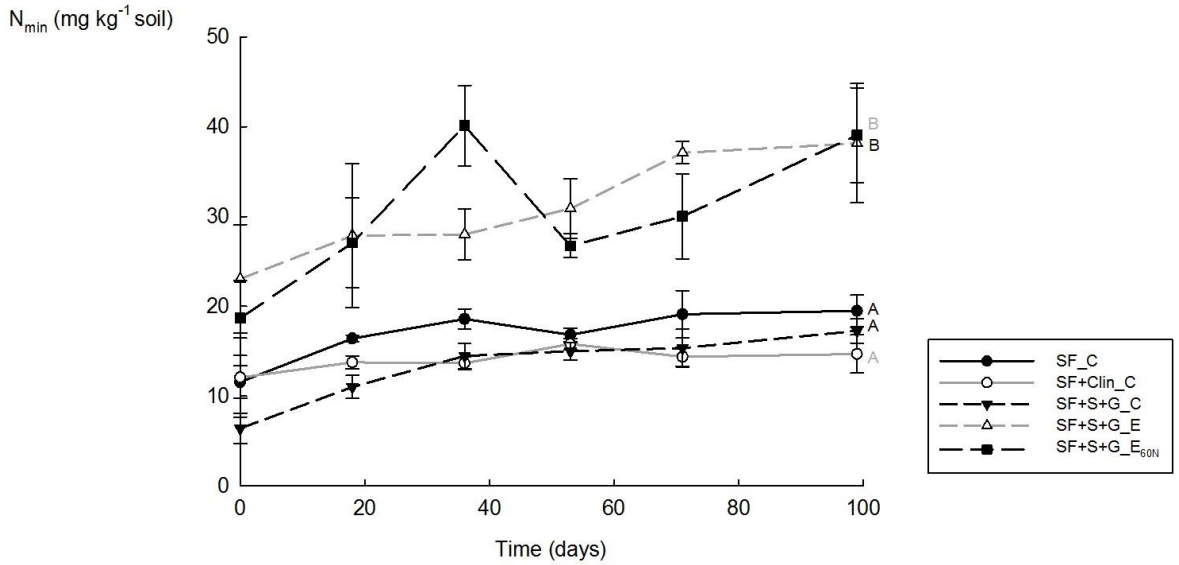


Figure 6.3 N mineralization (N_{min}) for composted solid fraction (SF) (SF_C), composted SF with clinoptilolite (SF+Clin_C), co-composted SF with straw and grass (SF+S+G_C), co-ensiled SF with straw and grass (SF+S+G_E) and co-ensiled SF with straw and grass with N fertilizer input (SF+S+G_E_{60N}). Error bars indicate standard deviations (n = 3). Treatments indicated with the same letter on day 100, are not significantly different (Scheffé test, $p < 0.05$).

6.4. Conclusion

Co-composting and co-ensiling are two promising valorization options for the SF of cattle slurry, resulting in different end products. Answering our four initial hypotheses, we found that: (1) Only a limited OM degradation was noticed when (co-)ensiling SF, resulting in a less stabilized product than with composting, prone to further decomposition once the silage is opened and applied to the soil. Soil amendment with co-ensilaged SF resulted in a higher N mineralization, NFRV and crop growth compared to (co-)composted SF, because of the higher NH_4^+-N concentrations in the silages. Moreover, with ensiling, more N and C can be added to the soil per unit of P (higher N/P and C/P ratios) than with composting. Therefore, silages are preferred for local use, while composted SF is better to export, as more P is concentrated in the composts per unit of fresh weight. (2) Co-composting using bulking agents enhanced the aeration and composting process, resulting in better sanitation which is important for export, but did not result in a more stable end product compared to composting pure SF without bulking agents. (3) Notwithstanding the high temperatures during composting SF, no increase in presence of X-TAS spores was observed in the study after composting, beneficial for use as bedding material. (4) Adding 2% clinoptilolite to the SF did not conserve more N during composting as was hypothesized. The second aim of this study was to compare different stability indicators for composts and silages. The stability indicators clearly differentiated the composts from the silages, related to the difference in OM degradation and N dynamics. The OUR was found to be less powerful to determine the stability of fresh, composted and ensiled SF.

CHAPTER 7

Collaborative forms of on-farm composting: Case studies

Abstract

Alternative, collaborative forms of on-farm composting might function as a lever to overcome some of the legislative, market and financial barriers. To gain more insights in the feasibility of some of these alternative production forms, three action research case studies were performed on different locations in Flanders, in collaboration with stakeholders and relevant policy advisors and implementers. Compost quality, production costs and applicable regulation per case were monitored. We examined two types of collaboration. In the first type, the on-farm production process was outsourced to a service provider (contractor), who was responsible for the set-up and monitoring of the on-farm composting process. In the second type, on-farm composting was performed by the collaboration between different farmers and nature conservationists, exchanging biomass and compost. The cases studies indicated that cooperation between different partners for the production of on-farm compost resulted in a better and economically more feasible composting process. Further, five general factors determining the choice of compost production set-up and the costs were illustrated: (1) available biomass, (2) local opportunities, (3) local compost requirements, (4) agreements made between partners and (5) facilities and capabilities of the partners. In conclusion, we formulated a set of suggestions for adjustments in legislation and policy, which could enhance chances for on-farm composting and farm compost application.

7.1. Introduction

On-farm composting is generally defined as an individual farmer recycling his or her own farmyard residues (e.g., crop residues, grass clippings, animal manure) into compost and using the compost on his or her own fields. However, despite the acknowledged benefits of compost use as soil improver and crop fertilizer, individual farmers rarely have the intention to produce compost themselves (Viaene et al., 2016b, described in detail in Chapter 2). An important barrier is the **complex and not stimulating legislation,** impeding farmers from composting. A particularly difficult requirement is the obligation to have an environmental license when using external biomass. Since the highly specialized nature of farms in Flanders, **the necessary feedstock composition for a good composting process is generally not available at one farm** and must be obtained via a third party. The requirements to obtain such a license can include a number of costly investments such as installing a concrete pad for composting and a system to capture and store run-off waste water. Furthermore, to produce a good quality compost, the compost should be monitored and managed, which requires specific tools and machines and hence a **financial investment**: an appropriate thermometer and a CO_2 sensor to monitor the process, a pile cover to avoid rainwater seeping into the compost, and a compost turning machine to aerate or moisture the compost. In addition to the financial investment, composting also requires a significant **investment of time of the farmer** to monitor the process and aerate the compost when necessary. Furthermore, a **lack of experience and knowledge** about the composting process and the regulations, together with the lack of stimulation by education and extension services might prevent farmers from composting farmyard residues. Hence, compost production in Flanders is currently dominated by professional, (semi-)industrial composting plants. However, farmers frequently note that local **compost supply for agricultural purposes is insufficient** and they experience a **variable and uncertain compost composition, quality and price** in the compost available from (semi-)industrial compost plants. Figure 7.1 repeats the revealed hindering factors from Chapter 2, highlighting the ones focused upon in more detail in this chapter.

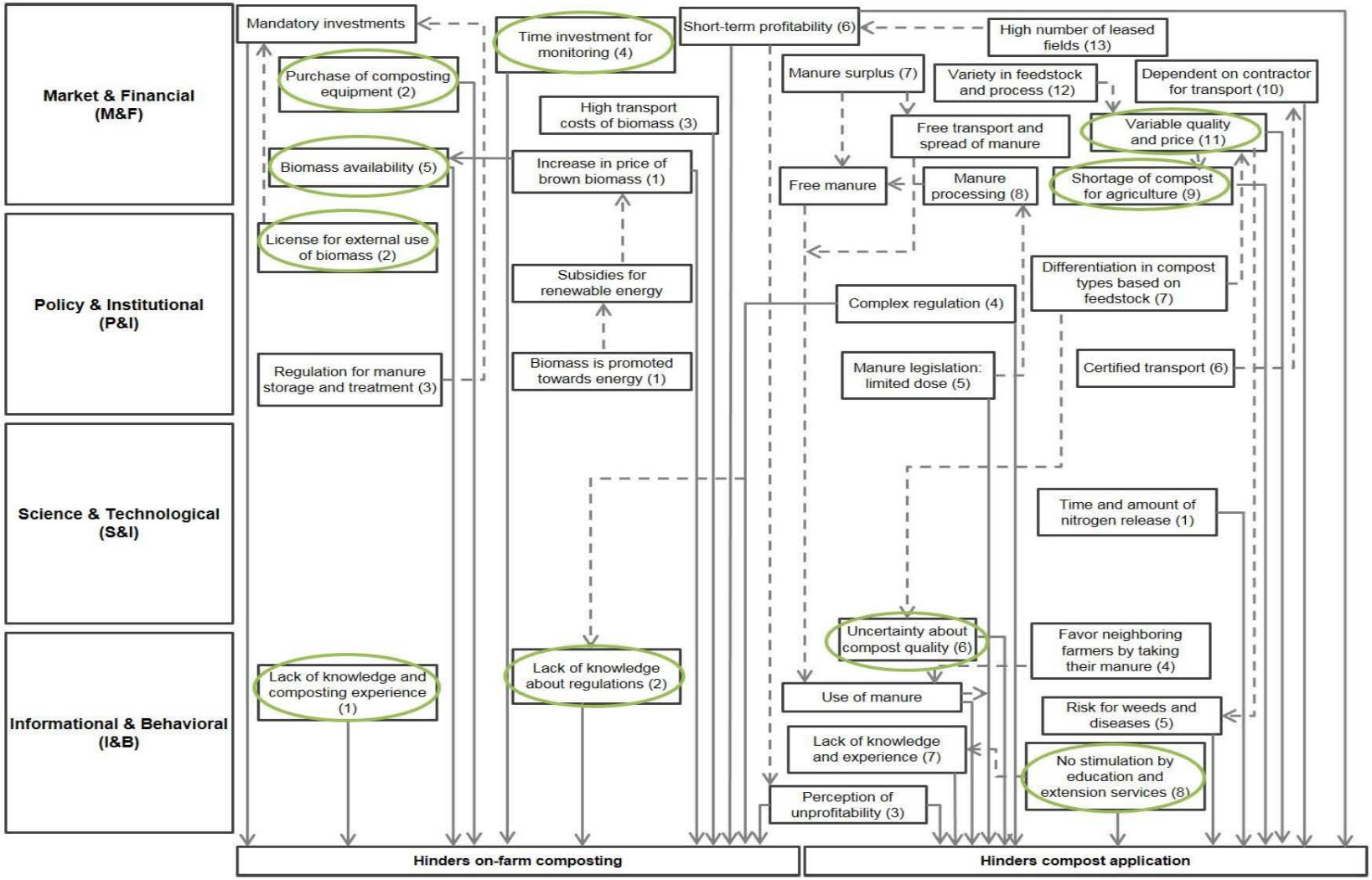


Figure 7.1 Overview of the different hindering factors to on-farm composting and compost application. The green circles are indicating the ones focused upon in Chapter 7. The numbers between brackets are corresponding to the number of hindering factors, as discussed in Chapter 2.

In this chapter we present the results of a set of action research cases on alternative, collaborative forms of on-farm composting as a means to overcome the above mentioned barriers and to better valorize the potential benefits of on-farm composting. For example, cooperation between farmers could lead to a higher availability of different types of biomass, improving the feedstock mixture and thus potentially a better compost quality. Furthermore, costs and efforts can be shared. The feasibility of several alternative collaboration forms was tested by performing three case studies on different locations in Flanders with different feedstock materials. Compost quality, production costs and applicable regulation per case were monitored in detail. To enhance the chances for these alternative production forms to be further adopted, the cases were set up, executed and discussed in collaboration with all involved stakeholders. They were also discussed with some relevant policy advisors and the Public Flemish Waste Agency (OVAM), a policy implementing agency.

We distinguished between two types of collaboration, each of them further adaptable to the available biomass type, local opportunities and requirements and facilities of the implementing stakeholders. In the first type, the on-farm production process was outsourced to a service provider, who was responsible for the set-up and monitoring of the on-farm composting process (*case study 1*). In the second type, on-farm composting was performed through the collaboration between different farmers and nature conservationists. In *case study 2*, two partners were collaborating, i.e. a farmer and a nature conservation organization, the former both executing the composting, providing part of the feedstock materials and being the end user of the compost, and the latter being responsible for the supply of part of the feedstock materials and the letting of the composting equipment. In *case study 3*, more than two partners (several farmers and a nature conservation organization) were collaborating in providing the feedstock materials and necessary equipment, performing the composting, and ultimately sharing the end product.

By studying these different types of cooperation between those stakeholders, we aimed at gaining more insight in: (1) the impact of the abovementioned barriers for these particular cases, and (2) the extent to which these collaborations enabled tackling these barriers. An overview of the three case studies is given in Table 7.1, together with the relevant barriers that we tried to tackle per case.

Table 7.1 Description of the three case studies, including the business as usual (BAU) and alternative scenario. GWC = green waste compost, FYM = farmyard manure.

Case study	BAU scenario	Alternative scenario				
		What	Used feedstock	Composter	Compost use	Tackled barriers
1. On-farm composting on a tree nursery by a service provider	Application of GWC: 30-35 t ha^{-1}, total of 800 t $year^{-1}$ Application of FYM: 20-25 t ha^{-1}, total of 200 t $year^{-1}$ Shredding of woody residues: 300 t $year^{-1}$	Co-composting of FYM, woody residues and additives 8 piles (LxHxW = 50x1.5 x3.5 m), ca. 2000 m^3 in total Turning events: 8	Proprietary: old (2013) and fresh (2014) woody residues from 2 years (pruning waste, market surplus): total of 600 t External: FYM from neighboring farmer: 200 t $year^{-1}$ External: bentonite from service provider to improve clay-humus complex: 480 kg	Service provider	1120 m^3 or 795.2 t On the own farm	Need for additional external feedstock Lack of experience and knowledge about composting process and regulation Time investment Financial investment for purchasing equipment Uncertain and variable GWC composition, quality and price Shortage in local compost supply for agricultural practices
2. On-farm composting with byproducts from nature reserves by hiring composting equipment (2 partners)	Stockpiling CFM without treatment: 200 m^3 $year^{-1}$	Co-composting of own byproducts + residues from nature management 1 pile (LxHxW = 55x1.8x4 m), ca. 400 m^3 in total Turning events: 8	Proprietary: FYM (75 m^3) and grass clipping (5 m^3) Proprietary: slurry (3000 L, mixture from cows and pigs) and water from a pond (3000 L) were added after 1 month composting External: grass clippings (212 m^3) and woody residues (80 m^3) from neighboring nature reserve	Farmer who hires monitoring equipment and turner for entire composting period	100 m^3 or 48.8 t On the own farm	Need for additional external feedstock Financial investment for purchasing equipment Uncertain and variable compost composition, quality and price Shortage in local compost supply for agricultural practices
3. On-farm composting with byproducts from nature reserves by hiring composting equipment (> 2 partners)	Transport of FYM to organic farmers (300 m^3 $year^{-1}$) + partly application on own fields	3 scenarios, 1 pile per scenario (LxHxW = 10x3x1.5 m), ca. 40-45 m^3 in total (A) Composting FYM (B) Co-composting FYM with hay from nature reserves (C) Co-composting FYM with pre-composted byproducts from nature reserves Turning events: 3	Proprietary: FYM External: old hay of grass, more coarse hay, wood chips and fresh grass clippings from neighboring nature reserve (A) 45 m^3 FYM (B) 17.5 m^3 FYM + 22.5 m^3 old hay from nature reserve (C) 17.5 m^3 FYM + 22.5 m^3 precomposted byproducts from nature reserve	(A) Farmer who hires turner when necessary (B) and (C) Nature conservationist who owns compost turner	(A)* 22.5 m^3 or 14.4 t (B)* 20 m^3 or 7.8 t (C)* 20 m^3 or 6.6 t *volumes were extrapolated to 150 m^3 compost for cost calculation On the own farm + neighboring organic farmers	Need for additional external feedstock (B and C) Lack of experience and knowledge about composting process and regulation (B and C) Time investment (B and C) Financial investment for purchasing equipment Uncertain and variable compost composition, quality and price Shortage in local compost supply for agricultural practices

7.2. Methodology

As researchers, we participated in the cases ourselves by e.g. bringing the stakeholders together, helping to select the appropriate feedstock materials, estimating the costs, etc. As we did not only observe and interpret the results, but were also involved in the process and negotiations, this type of research can be considered as action research. Each of the cases started with an individual in-depth interview of all stakeholders, firstly to identify the *Business As Usual* (BAU) characteristics, amounts, time of production and current valorization (and associated problems) of their byproducts. Furthermore, the requirements for compost application (quantity and quality), on-farm facilities for composting and knowledge about composting was questioned. In a second step, stakeholders were brought together to discuss the opportunities and best alternative scenario. Furthermore, arrangements were made about the costs and practicalities, e.g. type and amount of byproducts in order to have an optimal feedstock mixture for composting, means of transport for byproducts and compost, timing, location for the compost piles, equipment required to set up and monitor the piles, etc. See Table 7.1 for an overview of the case studies and the corresponding BAU and alternative scenarios. In addition, the applicable regulations for the specific organizational form were assessed.

Once all arrangements were made, the compost experiments could start. The separate feedstock materials, feedstock mixtures and composts were sampled for standard physico-chemical analyses as described in previous chapters (Table 1.1), in four replicates. This allowed to assess the compost quality and compare it with four different standards (see Chapter 1, Table 1.2). Pile temperature and moisture content were recorded to adjust the composting process if necessary. In order to enable a financial comparison, for each of the alternative scenarios, all costs related to the used feedstock materials (e.g., transport and pretreatment) and related to the on-farm compost production, transport and application were calculated and compared to a BAU scenario. All costs were calculated VAT excluded. To express the cost per tonne compost (on fresh matter (FM) and organic matter (OM) basis), we used the OM content of the compost (described in 7.3.1) and the amounts of feedstock and compost used in the different case studies (Table 7.1). For the costs linked to machine or material use, a mean cost was estimated including the labor costs of the operator and the operating (fuel), maintenance and depreciation costs of the used machine/material, using average machine/material capacities. An overview was made of fixed and variable standard costs, as well as other data (Table 7.9), based on own data, stakeholder consultation and/or

online data search. These values can be used to calculate other company-specific scenarios as well. After completing the composting process, a focus group was organized in which the results about the compost quality and cost analyses were disseminated, and the experiences of all stakeholders were shared. Furthermore, suggestions for improvement in terms of composting process, logistic arrangements, regulations, etc. were discussed. Finally, questions and recommendations regarding bottlenecks in and potential adaptation of current policies and institutional arrangements were deliberated with policy advisors and implementers. The specific details about the different cases are discussed in the sections below.

7.2.1. Case study 1

7.2.1.1. Context

SYLVA, a tree nursery located in Waarschoot, is a Belgian producer of forest and hedging plants. They annually produce >100 different varieties of forest and hedging plants on an area of 100 ha, of which 80 ha is planted. Agricultural practices with respect to the environment are an important corporate value, e.g. they use mechanical weed control and integrated pest management, are a participant of MPS (Environment Project Floriculture) and have flower meadows and insect hotels. Tree nurseries generate a considerable amount of byproducts throughout the year, especially pruning and sorting remains, e.g. saplings without straight stems (mainly between October and April), and market surpluses (mainly between May and October). When calculating the total production based on an average tree (two-year-old seedlings of birch), *SYLVA* produces around 10 t ha^{-1} $year^{-1}$, of which ca. 20% (2 t ha^{-1} $year^{-1}$) is lost by sorting. Together with the pruning residues and market surpluses, they have about 300 t of woody residues annually. Those residues consist of branches, stems, roots, some leaves or needles and contain a lot of C (Figure 7.2). Consequently, they can act as bulking agent (providing structure) and source of C for composting. For most tree nurseries, it is however too expensive to transport these surpluses (or have it picked up) to a professional composting facility, given the transport costs and gate fee between € 25-50 t^{-1} fresh material. Since the woody residues contain too much soil and sand, transport to a bioenergy plant is generally neither an option.

Figure 7.2 Woody residues from a tree nursery.

7.2.1.2. Business as usual scenario

In the BAU scenario, the company pays a contractor to shredder and occasionally incorporate the shredded wood on their fields, with risk for N shortage for the seedlings during further on-field degradation. Besides the high cost and risk for N shortage, sowing and planting machines can knot due to the coarse (> 4 cm) structure of the shredded wood. A finer shredding or sieving would increase the costs. Regular application of organic material to maintain or increase the soil organic matter content is very important for the nursery, especially because they are cultivating on a sandy soil in a catchment area for drink water production. Therefore, they annually apply around 30-35 t ha^{-1} (externally purchased) green waste compost in spring as basic fertilization on the fields where they sow. However, they, and other farmers, have some issues with the purchase of compost from professional compost facilities as explained in Section 7.1. On the other fields, the tree nursery applies farmyard manure (FYM) from a neighboring farmer (20-25 t ha^{-1}), who transports and spreads it for free. With this organic fertilization they stay below the fertilization limits for P_2O_5 and N. In conclusion, the **BAU scenario for the tree nursery is a combination of the application of purchased green waste compost and free FYM, and shredding and occasional application of own woody residues** (Table 7.1).

7.2.1.3. Alternative scenario

It could be favorable for the tree nursery to valorize their woody residues with FYM by on-farm composting, thereby producing their own compost. In that way, they process their byproducts, and produce compost that fulfills their requirements. Nevertheless, currently it

seems unfeasible to compost their selves because of the financial and time investments and the lack of knowledge about the composting process. Therefore, in the **alternative scenario, the on-farm production process was outsourced to a service provider, who was responsible for the set-up and monitoring of the on-farm composting of shredded woody residues and FYM at the tree nursery** (Table 7.1). The service provider, *Acterra*, is a private company, amongst others producing soil improvers by composting. Besides a long-term experience in composting, they own the required material to set up and monitor the composting process, such as temperature meters and a self-propelled compost turner. They have invested in a transport machine for their compost turner (Sandberger®, Figure 7.3), to have the opportunity to compost on other locations than their company. Furthermore, the compost turner is equipped with caterpillar tracks to minimize soil compaction, a system to supply water pumped over 300 m distance, and an automatic system to put on/off pile covers. On-farm composting was performed with a combination of approximately 600 t woody residues from two years (old and fresh residues) and 200 t FYM.

Figure 7.3 Compost turner of the service provider.

This type of cooperation would address a number of barriers to on-farm composting (Table 7.1). By co-composting FYM from a neighboring farmer, the tree nursery has additional N-rich feedstock to mix with the C-rich woody residues, thereby improving the feedstock mixture for composting and produce a more nutrient-rich compost. Further, the tree nursery would have to invest significantly less time in producing the compost and would not need to purchase the required equipment. Additionally, this would circumvent the informational hindering factor of lack of knowledge about the composting process and experience in on-

farm composting. Lastly, the tree nursery will be guaranteed about the composition of the compost and the availability when they need it.

The composting process of 79 days started on July 16th 2014 and lasted until October 2nd 2014. The compost piles are illustrated in Figure 7.4. Separate feedstock materials and three out of eight piles were sampled (in four replicates) at the start after turning once, and the end of the composting process. The results of the farm compost were compared with the Federal standards, the purchased green waste compost and FYM.

Figure 7.4 Compost piles at the tree nursery.

7.2.2. Case study 2

7.2.2.1. Context

Widar (Merksplas, Belgium) is a social community (non-profit organization) where disabled people, supervisors and their family live and work together. They produce vegetables, fruit and dairy and meat products, all according to the biodynamic principles. The byproducts include FYM (200 m^3 $year^{-1}$), grass clippings (40-60 m^3 ha^{-1}), wood chips and shavings (also occasionally from the municipality) and crop residues.

7.2.2.2. Business as usual scenario

The **BAU scenario** for *Widar* is rather dynamic and therefore difficult to describe. The farm is developing continuously and adapting the valorization of their farm residues and fertilization to the opportunities at the time. In the beginning, the farm had an area of only 1 ha for horticulture and crop residues were properly composted by hand. As the farm expanded, they included animal farming. The FYM (annually ca. 200 m^3) was put on piles

with a manure spreader (without covering), occasionally crop residues were mixed with the FYM. However, the farmer had not enough time to compost properly, meaning that the composting process (temperature, CO_2, moisture content) is not monitored (Table 7.1). Furthermore, the compost is not aerated with a compost turner when necessary, and not covered with a pile cover to prevent the infiltration of rainwater, possibly causing anoxic conditions leading to a suboptimal compost quality. Additionally, due to this passive composting process, it lasts for half a year to stabilize the compost, after which they apply around 15 t ha^{-1} on their own pastures. However, to optimize the composting process and to invest themselves in the purchase of composting equipment, would be a huge investment in time and budget for *Widar*. We chose to compare the alternative scenario with **storage (on piles with a manure spreader) and spreading of FYM**.

7.2.2.3. Alternative scenario

Currently, *Widar* has the opportunity to use grass clipping from nature reserves, which is interesting to increase the C content of their soils. *Natuurpunt*, the largest nature conservation association in Flanders, is searching for cheaper opportunities for the off-take of their residues from nature management, given the often high transport costs and the gate fee to be paid for the current off-take at professional composting or digesting facilities. Therefore, **locally available nature residues** (grass clippings and wood chips from birch and pine) were transported to *Widar*, to **co-compost with the FYM**, thereby improving the feedstock mixture and composting process in the alternative scenario. *Widar* **hired the composting equipment** for the entire composting period, i.e. a thermometer, CO_2 sensor, pile cover and compost turner (Ménart type SP4300) from *Natuurpunt*, (Figure 7.5). Fresh grass clippings were picked up and directly transported to Widar (max. 3 km) by *Natuurpunt*, three days before the set-up of the compost pile. Grass clippings mainly consisted of *Pitrus* (80%). Wood chips were transported over 18 km by a contractor. *Natuurpunt* helped setting-up the compost pile at the farm.

This type of cooperation would address a number of barriers to on-farm composting (Table 7.1). Bringing feedstock materials and letting composting equipment to *Widar*, in exchange for processing the feedstock materials from *Natuurpunt*, could be a win-win case for both parties. The individual farmer gets additional biomass for free from nearby nature reserves in return for valorizing them. Furthermore, the financial investment for composting equipment is strongly reduced by hiring it. The farmer has also more control over the compost availability, composition and quality, through adjusting and optimizing the process.

The lack of compost for agricultural purposes is especially important for organic and biodynamic farming, because they are limited to green waste compost from biological feedstock material.

The composting process started on August 18th 2014 and lasted until October 13th 2014 (57 days). Separate feedstock materials and compost pile were sampled at the start and the end of the composting process and compared with the Federal compost standards and initial FYM.

Figure 7.5 Turning of the compost (left) and covered compost pile (right).

7.2.3. Case study 3

7.2.3.1. Context

The 'nature farm' *Het Bolhuis* (Molenstede, Belgium) produces organic meat (beef, pork, lamb and chicken) from rare breeds such as *Kempische runderen* and *Ardense Voskoppen*. Het *Bolhuis* has excess FYM (around 300 m^3 $year^{-1}$) available between February and April that cannot be used on own fields. The farm has a parcel user license with *Natuurpunt* (i.e. they are mowing and let their animals graze on certain parcels of *Natuurpunt* at no cost but are subject to certain conditions set by *Natuurpunt*). The hay used as bedding material in the stables of *Het Bolhuis* is originating from those nature reserves, hence, free from pesticides, antibiotics, etc. Therefore, the resulting FYM containing this hay, is appreciated by nearby organic farmers as organic fertilizer. In addition, in the region there is a high demand for organic fertilizers in arable farming and horticulture to increase the SOC content and improve soil biology. However, either these farmers compost themselves, but do it in a passive way as described in case study 2, or they do not have the capacity and/or knowledge to compost and apply the FYM as such. Quality is therefore not always ideal, with potentially

relatively high weed pressure due to weed seeds in the FYM. We chose to compare the BAU and alternative scenario from the viewpoint of *Het Bolhuis*.

7.2.3.2. Business as usual scenario

Currently, the **FYM from *Het Bolhuis* is put on big heaps on a concrete floor without cover or treatment for 2-3 months, prior to being transported to different organic farms in the region** (Table 7.1). The FYM itself is for free, but the end-users pay for the transport. On the one hand, *Het Bolhuis* wants to upgrade the quality and reduce the environmental impact of storing the FYM by composting, and on the other hand, the organic farmers want to reduce transport costs and apply high-quality compost rather than pure FYM.

7.2.3.3. Alternative scenario

We tested three alternative scenarios for on-farm composting of cattle FYM, i.e. (A) composting of pure FYM, (B) co-composting of FYM with hay of grassland management from adjacent nature reserves (eight months old, stored in bales) (44:56% v/v), and (C) co-composting of FYM with pre-composted hay of grassland management from adjacent nature reserves comprising feedstock mixture of old hay, more coarse hay, wood chips and fresh grass clippings (44:56% v/v) (Table 7.1 and Figure 7.6). In all scenarios, a thermometer and CO_2 sensor were bought and the pile was monitored by *Het Bolhuis*. In scenario A, the compost turner was hired from *Natuurpunt* (compost turner from Ménart type SP4300), and turning was performed by *Het Bolhuis*, while in scenarios B and C, turning was performed for free by *Natuurpunt*, in exchange for the off-take of their byproducts. Furthermore, in scenarios B and C, the addition of more C-rich materials to the FYM, could reduce nutrient losses during composting and improve compost quality. The hay from different areas was collected in bales on a temporary storage location. Since the transport costs for the hay to *Het Bolhuis* are high, it was tested in scenario C whether it was feasible for *Natuurpunt* to prior to the transport, compost the hay with other byproducts to reduce its volume and thus the transport costs. Moreover, also this compost, consisting purely of plant-based byproducts, could be sold to nearby farmers.

Scenario A could provide a solution for some bottlenecks of both parties: (i) the financial investment for *Het Bolhuis* for purchasing composting equipment by hiring it, and (ii) the shortage in compost supply and uncertain and variable compost composition, quality and price for the buyers. The cooperation between different farmers and *Natuurpunt* in scenarios

B and C could tackle the same barriers as described in case study 2, plus the lack of experience and knowledge and time investment, since *Natuurpunt* is performing the composting process (Table 7.1).

The composting process started on April 14th 2015 and lasted until June 10th 2015 (56 days). Separate feedstock materials and compost piles were sampled at the start and the end of the composting process and compared with each other, the Federal compost standards and initial FYM.

Figure 7.6 Set-up of the piles (left) and turning of the compost (right).

7.3. Results

7.3.1. Compost quality

7.3.1.1. Case study 1

The old shredded wood residues had a much lower OM and N content and C/N and C/P ratio than the freshly shredded ones (Table 7.2), i.e. they were already more degraded and contained less easily degradable C. There was some variation in feedstock mixtures of the three piles (Table 7.2), related to the proportion of manure/wood in the samples at the start. After composting, those differences were negligible, therefore, the average of 12 samples was calculated to have a general idea about the compost quality. Pile temperatures reached 70°C shortly after the start of the experiment and were above 55°C for one month (data not shown), indicating destruction of pathogens and weed seeds (Table 7.2). The piles were turned eight times in total.

The OM content of the farm and green waste compost (Table 7.3) was lower compared to the uncomposted FYM (65.5% of DM), since OM degradation occurs during composting.

Consequently, FYM contained more labile C and will further degrade after soil incorporation, also indicated by reported OUR and biodegradation potentials of FYM (Viaene et al., 2016a). The OM content of the farm compost (10.2% of FM) was below Federal standards, explained by a high soil load of the feedstock mixture (Table 7.3). As a matter of fact, during the set-up of the piles, a considerable amount of soil was shoveled up, confirmed by the lower OM content of the feedstock mixtures compared to the separate feedstock materials (Table 7.2). The OM content of the purchased green waste compost (13.0% of FM) did not meet the Federal OM standard. The dry matter (DM) content increased during composting and both composts met the Federal standards and were considerably drier than FYM (22.4%). pH of both composts were in line with the standards (Table 7.3). EC of the farm compost (667 µS cm^{-1}) was higher than the green waste compost (220 µS cm^{-1}), related to the addition of FYM (higher K content), however, no problems with salinity are expected (e.g. a max. value of 750 µS cm^{-1} for use in potting media). C/N and C/P ratios were considerably higher for the green waste compost compared to the farm compost (Table 7.3), because the latter contained more nutrients (N, P, K) due to the addition of FYM and is more stabilized (more C losses). The C/P and N/P ratios, N and P content of the uncomposted FYM were higher than the farm compost, but there was no difference in C/N ratio (Table 7.3). The temperatures of the farm compost lowered to 35-40°C at the end of the composting process, indicating a decrease in degradation activity and increase in stability. This was confirmed by the NO_3^--N/NH_4^+-N ratio > 1 and OUR ≤ 5 mmol kg^{-1} OM h^{-1} (Table 7.3). In contrast, the green waste compost had a NO_3^--N/NH_4^+-N ratio < 1, pointing towards unstable compost, but an OUR ≤ 5 mmol kg^{-1} OM h^{-1}, indicating very stable composts according to the Federal standards. However, the OUR measures the apparent stability under the current microbial activity of the lab conditions and can be limited by other factors such as mineral N shortage.

Table 7.2 Physico-chemical composition of the feedstock materials and feedstock mixtures (mean ± standard deviation, n = 4). FYM = farmyard manure, OM = organic matter, DM = dry matter, FM = fresh matter.

	Feedstock			Feedstock mixtures		
	Fresh wood residues	Old wood residues	FYM	Pile 1	Pile 2	Pile 3
OM content (% of DM)	62.5 ± 16	20.4 ± 7.4	65.5 ± 11.1	27.2 ± 5.6	32.8 ± 6.8	44.8 ± 12.1
DM content (% of FM)	67.4 ± 7.8	52.8 ± 8.4	22.4 ± 2.3	51.4 ± 5	54.4 ± 2.3	52.5 ± 8.7
N (g kg^{-1} DM)	9.5 ± 1.4	5.8 ± 1.4	23.2 ± 3.3	8.3 ± 1.4	9.1 ± 2.7	9.3 ± 2.0
P (g kg^{-1} DM)	1.6 ± 0.2	1.5 ± 0.3	4.6 ± 0.6	1.9 ± 0.3	1.5 ± 0.1	3.0 ± 0.2
C/N (-)	37 ± 7	19 ± 4	16 ± 1	18 ± 4	21 ± 6	27 ± 5
C/P (-)	222 ± 54	78 ± 23	79 ± 4	79 ± 17	118 ± 17	102 ± 43
NO_3^--N (mg kg^{-1} DM)	nda	nda	nda	90 ± 16	24 ± 7	39 ± 11
NH_4^+-N (mg kg^{-1} DM)	nda	nda	nda	53 ± 30	400 ± 86	554 ± 229
NO_3^--N / NH_4^+-N (-)	nda	nda	nda	2.495 ± 2.170	0.062 ± 0.027	0.083 ± 0.049
Fresh bulk density (kg m^{-3})	nda	nda	nda	513 ± 48	257 ± 49	428 ± 31
Germinated weeds (number L^{-1})	nda	nda	nda	1.3 ± 0.6	0 ± 0	0.3 ± 0.6

Table 7.3 Physico-chemical composition of the cattle farmyard manure (FYM), purchased green waste compost (GWC) and farm compost (mean ± standard deviation, n = 4). Parameters indicated with the same letters are not significantly different (Scheffé test, $p < 0.05$). OM = organic matter, DM = dry matter, FM = fresh matter.

	FYM		GWC		Farm compost		*Federal norm*
OM content (% of DM)	65.5 ± 11.1	b	19.6 ± 3.6	a	16.8 ± 5.6	a	
OM content (% of FM)	14.5 ± 1.5	a	13.0 ± 2.6	a	10.2 ± 3.3	a	*> 16*
DM content (% of FM)	22.4 ± 2.3	a	66.5 ± 1.3	c	61.2 ± 2.9	b	*> 50*
N (g kg^{-1} DM)	23.2 ± 3.3	c	4.6 ± 0.2	a	6.5 ± 1.3	b	
P (g kg^{-1} DM)	4.6 ± 0.6	c	1.0 ± 0.0	a	1.7 ± 0.2	b	
C/N (-)	16 ± 1	a	24 ± 5	b	14 ± 4	a	
C/P (-)	79 ± 4	b	114 ± 17	c	55 ± 16	a	
N/P (-)	5.1 ± 0.1	b	4.8 ± 0.4	ab	3.8 ± 0.7	a	
NO_3^--N (mg kg^{-1} DM)	nda		1 ± 0	a	36 ± 22	b	
NH_4^+-N (mg kg^{-1} DM)	nda		10 ± 7	a	5 ± 2	a	
NO_3^--N / NH_4^+-N (-)	nda		0.158 ± 0.094	a	8.706 ± 5.791	b	
Fresh bulk density (kg m^{-3})	nda		517 ± 26	a	710 ± 25	b	
Germinated weeds (number L^{-1})	nda		0 ± 0	a	0 ± 0	a	*max. 1*
pH-H_2O (-)	nda		7.76 ± 0.08	a	8.07 ± 0.23	b	*6.5 - 9.5*
Electrical conductivity (µS cm^{-1})	nda		220 ± 40	a	667 ± 270	b	
Oxygen uptake rate (mmol kg^{-1} OM h^{-1})	nda		5.0 ± 0.9	a	4.8 ± 1.4	a	*< 15*
K (g kg^{-1} DM)	nda		1.8 ± 0	a	4.4 ± 1.3	b	

7.3.1.2. Case study 2

Pile temperatures raised until 65°C at the start and were above 55°C for 22 days (data not shown), indicating destruction of pathogens and weed seeds (Table 7.4). After one month, the pile became too dry, consequently slurry and water was added, which elevated the CO_2 concentrations immediately, indicating oxygen shortage. Therefore, it was concluded that we added too much moisture to the pile in one action. Piles were turned seven times in total.

The OM content of the feedstock mixture (36% of DM) was low compared to the separate feedstock materials (54-95% of DM), possibly from shoveling up a lot of soil with the set-up of the compost pile. The OM content of the final compost was 21% (of FM), well above Federal and *Vlaco* standards. The OM content increased compared to the start: on the one hand, C was lost by CO_2 loss, but on the other hand, C (and nutrients) were added by the extra addition of slurry after one month. The DM content of 51%, pH and EC were in line with the standards (Table 7.4). The C/N ratio of the farm compost (17) was comparable to average values for green waste compost. C/P and N/P ratios of the compost (Table 7.4) were lower than the averages for green waste compost related to the addition of FYM, but higher than the average ratios of vegetable, fruit and garden waste compost. K content was between averages for green waste compost and vegetable, fruit and garden waste compost (Table 7.5). The farm compost could not yet be considered stable after two months, based on the pile temperatures (53°C at the end), the wood chips that were still visible and the NO_3^--N/NH_4^+-N ratio < 1. In contrast, the OUR < 5 mmol kg^{-1} OM h^{-1} was pointing towards very stable compost according to Federal standards, explained by the apparent stability in section 7.3.1.1.

7.3.1.3. Case study 3

Pile temperatures were lowest for the pile with only FYM (max. 61°C), followed by the pile where old hay was added (max. 64°C) and highest for the pile where pre-composted byproducts were added (max. 71°C). Maximum two weeds germinated after 21 days of incubation, indicating pile temperatures were sufficient to destroy weed seeds. All piles were turned three times in total.

Table 7.4 Physico-chemical composition of the cattle farmyard manure (FYM) from *Widar*, grass clippings from *Widar* and *Natuurpunt*, wood chips from *Natuurpunt*, feedstock mixture at the start and compost at the end of the composting process (mean ± standard deviation, n = 4). OM = organic matter, DM = dry matter, FM = fresh matter.

	FYM *Widar*	Grass clippings *Widar*	Grass clippings *Natuurpunt*	Wood chips *Natuurpunt*	Feedstock mixture	Compost
OM content (% of DM)	54.1 ± 17.5	92.3 ± 1.0	82.6 ± 17.0	95.3 ± 1.1	35.7 ± 4.0	41.2 ± 5.0
OM content (% of FM)	13.7 ± 2.6	37.5 ± 6.5	24.2 ± 3.2	31.7 ± 0.8	15.8 ± 1.8	20.8 ± 2.0
DM content (% of FM)	26.4 ± 4.6	40.7 ± 7.2	30.1 ± 5.9	33.3 ± 0.9	44.4 ± 0.8	50.7 ± 4.5
N (g kg^{-1} DM)	21.1 ± 2.5	14.6 ± 0.7	13.2 ± 2.5	8.2 ± 0.6	11.0 ± 0.9	14.0 ± 3.3
P (g kg^{-1} DM)	5.1 ± 1.4	2.7 ± 0.5	1.9 ± 0.5	0.5 ± 0.1	2.8 ± 0.1	3.2 ± 0.2
C/N (-)	14 ± 3	35 ± 1	35 ± 3	65 ± 5	18 ± 2	17 ± 3
C/P (-)	61 ± 23	196 ± 37	254 ± 31	1026 ± 168	72 ± 7	72 ± 12
N/P (-)	4.3 ± 1.1	5.6 ± 1.3	7.4 ± 1.1	15.7 ± 2.0	4.0 ± 0.2	4.4 ± 1.1
NO_3^--N (mg kg^{-1} DM)	nda	nda	nda	nda	40 ± 21	106 ± 40
NH_4^+-N (mg kg^{-1} DM)	nda	nda	nda	nda	512 ± 50	490 ± 73
NO_3^--N / NH_4^+-N (-)	nda	nda	nda	nda	0.079 ± 0.045	0.218 ± 0.075
Fresh bulk density (kg m^{-3})	nda	nda	nda	nda	nda	488 ± 46
Germinated weeds (number L^{-1})	nda	nda	nda	nda	9 ± 11	0 ± 0
pH-H_2O (-)	nda	nda	nda	nda	nda	7.65 ± 0.30
Electrical conductivity (µS cm^{-1})	nda	nda	nda	nda	nda	975 ± 98
Oxygen uptake rate (mmol kg^{-1} OM h^{-1})	nda	nda	nda	nda	nda	3.2 ± 1.1
K (g kg^{-1} DM)	nda	nda	nda	nda	nda	10.0 ± 0.5

The old hay, used in scenario B, had a higher OM and nutrient (N, P, K) content (on DM), and lower C/N ratio than the pre-composted byproducts, used in scenario C (Table 7.5). There was no difference in fresh bulk density between old hay and pre-composted byproducts as also other byproducts were added to the hay to compost. The OM content (on fresh weight) was considerably higher than the Federal and *Vlaco* standards in scenarios B and C and was highest in scenario C, but was below the standards in scenario A. The DM content of all composts did not meet the Federal standards. Total N, P and K content was lower in scenario C compared to the other scenarios. Scenario A had respectively lower and higher C/N and C/P ratios than scenarios B and C. The fresh bulk density was lowest for scenario C (330 kg m^{-3}) and almost twice as high for scenario A. All composts had a pH and EC in line with the Federal standards, and was highest for scenario A (Table 7.5). The pH was somewhat higher than average *Vlaco* composts. All composts could be considered stable according to the OUR < 15 mmol kg^{-1} OM h^{-1}. However, as was the case in case study 1 and 2, the other stability parameters indicated that the composts were not yet stabilized after two months, pile temperatures were still above ambient air temperature (47, 59 and 62°C for scenario A, B and C, respectively) and NO_3^--N/NH_4^+-N ratios < 1 (Table 7.5). There was no difference in NO_3^--N/NH_4^+-N ratio (Table 7.5) among the three scenarios, however scenario A contained less NO_3^--N and had a higher OUR than scenario B, indicating less stabilization.

7.3.2. Costs

7.3.2.1. Case study 1

In the BAU as well as in the alternative scenario, the woody residues must be shredded, which implies an extra cost of € 13 t^{-1} wood (€ 150 h^{-1}, 30 m^3 h^{-1} using a professional wood shredder).

Table 7.5 Physico-chemical composition of the cattle farmyard manure (FYM), old hay of grass and pre-composted byproducts from adjacent nature reserves, and compost at the end of the composting process for scenario A (composted FYM), B (co-composted FYM + old hay) and C (co-composted FYM + pre-composted byproducts from nature reserves) (mean ± standard deviation, $n = 4$). Parameters for old hay and pre-composted byproducts indicated with an asterisk are significantly different (*t*-test, $p < 0.05$). Compost parameters indicated with the same letters are not significantly different (Scheffé test, $p < 0.05$). OM = organic matter, DM = dry matter, FM = fresh matter.

	Feedstock materials			Scenarios on-farm compost					
	FYM	Old hay	Pre-composted byproducts	A		B		C	
OM content (% of DM)	59.2 ± 11.6	84.3 ± 3*	62.8 ± 3.2*	51.6 ± 4.9	a	53.4 ± 3.9	a	50.4 ± 3.2	a
OM content (% of FM)	15 ± 3.5	27.1 ± 6.1	24.7 ± 1.0	14.5 ± 0.5	a	20.3 ± 1.2	b	24.8 ± 1.8	c
DM content (% of FM)	25.8 ± 6.8	32.0 ± 6.7	39.4 ± 1.7	28.2 ± 2.2	a	38.3 ± 3.9	b	49.3 ± 0.9	c
N (g kg^{-1} DM)	19.9 ± 6.9	24.8 ± 4.2*	14.1 ± 1.3*	26.8 ± 2.7	b	26.2 ± 3.2	b	20.6 ± 1.3	a
P (g kg^{-1} DM)	3 ± 0.6	3.2 ± 1.2*	1.6 ± 0*	4.8 ± 0.5	b	4.4 ± 0.3	b	3.2 ± 0.1	a
C/N (-)	17 ± 4	19 ± 3*	25 ± 2*	11 ± 1	a	12 ± 2	ab	14 ± 1	b
C/P (-)	109 ± 10	164 ± 61	225 ± 14	60 ± 6	b	68 ± 5	a	87 ± 6	a
N/P (-)	6.6 ± 1.8	8.3 ± 1.8	9.1 ± 0.8	5.6 ± 0.8	a	6.0 ± 0.6	a	6.4 ± 0.5	a
NO_3^--N (mg kg^{-1} DM)	nda	nda	nda	43 ± 3	a	169 ± 85	b	93 ± 27	ab
NH_4^+-N (mg kg^{-1} DM)	nda	nda	nda	4326 ± 2367	a	1823 ± 2155	a	702 ± 71	a
NO_3^--N / NH_4^+-N (-)	nda	nda	nda	0.008 ± 0.001	a	0.122 ± 0.178	a	0.133 ± 0.041	a
Fresh bulk density (kg m^{-3})	399 ± 69	132 ± 29	170 ± 16	641 ± 21	c	391 ± 27	b	330 ± 18	a
Germinated weeds (number L^{-1})	nda	nda	1 ± 1	1 ± 1	a	2 ± 0	a	2 ± 1	a
pH-H_2O (-)	8.25 ± 0.44	nda	6.99 ± 0.17	9.1 ± 0.2	b	8.8 ± 0	ab	8.6 ± 0.1	a
Electrical conductivity (µS cm^{-1})	1197 ± 36	nda	196 ± 32	2219 ± 205	b	1625 ± 119	a	1328 ± 79	a
Oxygen uptake rate (mmol kg^{-1} OM h^{-1})	nda	nda	8.9 ± 2.2	13.5 ± 2.8	b	6.9 ± 1.0	a	10.1 ± 2.4	ab
K (g kg^{-1} DM)	nda	17 ± 7.2*	7.7 ± 0.3*	23.1 ± 2.1	b	22.0 ± 0.4	b	18.3 ± 0.8	a

In the **BAU scenario**, *SYLVA* pays € 2.5 t^{-1} green waste compost, which is transported for € 2.5 t^{-1} and spread for € 2.7 t^{-1} by a contractor (€ 50 h^{-1}), hence making the total cost of € 7.7 t^{-1} applied green waste compost. Costs for transport and spreading of FYM are zero. The cost for shredding and spreading the wood shavings by a contractor are € 16.0 t^{-1} wood (Table 7.6).

In the **alternative scenario**, the compost piles were set-up (total of 15 h for 8 piles at € 54.0 h^{-1}). Bentonite was added to the woody residues and FYM, to improve the formation of a clay-humus complex (€ 1.00 kg^{-1}). Transport of the necessary machinery to *SYLVA* amounted to 2x € 250. During the composting process, piles were monitored and turned eight times (1.5 h per event) at € 500 per event (transport + turning). This amounted to a total cost of € 7.3 t^{-1} compost, for producing 800 t compost. Adding the cost for shredding wood and spreading the compost by a contractor (at € 2.7 t^{-1}), this gives a total cost of **€ 23.3 t^{-1}** compost (Table 7.6).

If *SYLVA* would perform the composting process itself, the production cost would be € 8.1 t^{-1} compost (including purchase of composting equipment and time investment of set-up and monitoring, using general costs described in Table 7.9, with a compost turner from € 25,000 and a labor cost of € 25 h^{-1}), which is ca. € 0.8 t^{-1} more expensive compared to the production cost of € 7.3 t^{-1} when the composting process is executed by *Acterra*.

Table 7.6 Cost calculation for the business as usual (BAU) and alternative scenario for case study 1 (€ $tonne^{-1}$ end product, left column), based on the amount of end product (FM) produced in this particular case study (t $year^{-1}$, right column). GWC = green waste compost, FYM = farmyard manure, OM = organic matter, FM = fresh matter.

	BAU scenario			Alternative scenario	
	Cost (€ t^{-1})	Amount (t y^{-1})		Cost (€ t^{-1})	Amount (t y^{-1})
GWC			*Farm compost*		
Purchase	2.5	800	Shredding wood	13.3	600
Transport	2.5	800	Production	7.3	800
Spread	2.7	800	Spread	2.7	800
Total cost on FM base	7.7	800	Total cost on FM base	23.3	800
Total cost on OM base	59.2	104	Total cost on OM base	228.4	82
FYM					
Purchase	0	200			
Transport	0	200			
Spread	0	200			
Total cost on FM base	0	200			
Total cost on OM base	0	29			
Woody residues					
Shredding	13.3	300			
Spread	2.7	300			
Total cost on FM base	16.0	300			
Total cost on OM base	40.0	120			

7.3.2.2. Case study 2

In both scenarios we disregarded the costs for loading the manure from the stables to the storage place. In the **BAU scenario** 200 m³ or 100 t FYM is piled and spread with a manure spreader from *Widar* for a total cost of **€ 1.7 t^{-1}** FYM (Table 7.7).

In the **alternative scenario**, *Widar* could hire the necessary composting equipment from *Natuurpunt* for € 75 per week, consequently for eight weeks composting this is a total cost of € 600. Setting-up (14 h, 30 m^3 h^{-1}), turning (8 times, 0.5 h per pile, except for the first time when it took 1.5 h because the pile was too big) and monitoring (16 times, 0.3 h per event) the pile, resulted in a total cost of € 18.0 t^{-1} compost. In contrast to the other cases, we used a labor cost of only € 10 h^{-1} instead of € 25 h^{-1}, because *Widar* is subsidized by the government. When adding the cost for spreading by a contractor (€ 2.7 t^{-1}), the total cost is **€ 20.7 t^{-1}** compost (Table 7.7).

We compared the alternative scenario with a theoretical scenario of on-farm composting of FYM and feedstock from *Natuurpunt*, but when buying the composting equipment instead of hiring it. Hence, in this theoretical scenario, *Widar* would have extra costs from the purchase of composting equipment and the labor costs (from set-up and monitoring), using the general costs described in Table 7.9, with a compost turner of € 25,000. For composting the 200 m³ annual available FYM with 200 m³ grass clippings, resulting in a production of 120 t compost, this would result in a total cost of € 26 t^{-1} compost, which is more expensive than the scenario in which the composting equipment is hired.

Table 7.7 Cost calculation for the business as usual (BAU) and alternative scenario for case study 2 (€ $tonne^{-1}$ end product, left column), based on the amount of end product (FM) produced in this particular case study (t $year^{-1}$, right column). OM = organic matter, FM = fresh matter.

	BAU scenario		Alternative scenario	
	Costs (€ t^{-1})	Amount (t y^{-1})	Costs (€ t^{-1})	Amount (t y^{-1})
Piling	0.8	100	18.0	49
Spread	0.9	100	2.7	49
Total cost (FM base)	1.7	100	20.7	49
Total cost (OM base)	100.7	14	99.5	10

In addition, we also calculated the difference in costs for *Natuurpunt* for two scenarios respectively, (i) valorizing the grass clippings on a local scale by transporting them to a local farmer and helping by setting-up the compost pile, and (ii) transporting them to a compost facility at a distance of 10 km.

The cost for *Natuurpunt* to transport the grass clippings and wood shavings to Widar (max. 3 km) and putting the feedstock on piles is € 6 m^{-3} grass clippings. In this cost, the rent obtained from *Widar* for using the compost equipment of € 75 $week^{-1}$, was already deducted. The alternative is to transport the grass clippings to a compost facility, which costs € 5.6 m^{-3} plus a gate fee of € 35 t^{-1}. Converting the gate fee to tonnes (using a bulk density between 40-300 kg m^{-3} depending on the moisture content), the total cost is between 7 and 16 € m^{-3}, which is thus more expensive than transporting the grass clippings to a local farm.

7.3.2.3. Case study 3

In all scenarios we disregarded the costs for loading the manure from the stables to the storage place. In the **BAU scenario**, the cost for the livestock farmer is estimated to be **€ 2.7 t^{-1}** for spreading part of the manure on his own fields. The cost is **zero** if the manure is traded to other farmers, because they are paying for the transport of the manure to their farms.

In **alternative scenario A**, when composting annually 300 m³ FYM, resulting in 150 m³ compost, the total cost is € 4.9 m^{-3} compost or **€ 7.7 t^{-1}** compost (when using the measured bulk density of 641 kg m^{-3} compost; Table 7.8). In this total cost, a labor cost of € 25 h^{-1} is taken into account and the following operational costs (based on Table 7.9) were included: the costs for transportation by a contractor (1 km distance to the field where the composting takes place), set-up of the pile, turning with a hired compost turner (€ 55 h^{-1}) and monitoring (buying 2 compost covers and a thermometer). Hence, *Het Bolhuis* should ask a minimum price of € 7.7 t^{-1} to the compost users to avoid economic losses. In case part of the compost is for own use, the cost for paying a contractor to spread this compost on his own fields needs to be added, amounting the total cost to € **10.4 t^{-1}** compost (Table 7.8).

In **alternative scenario B**, when composting annually 300 m³ FYM, 300 m³ hay from *Natuurpunt* is needed, which is transported for free by *Natuurpunt* to the farm. In this scenario, the farmer buys the compost covers and thermometer, and pays the transport from the manure to the field. However, turning costs and using the compost turner is for free, in exchange for taking over the hay from *Natuurpunt*. This gives a total cost of € 2.2 m^{-3} compost or **€ 5.5 t^{-1}** compost (when producing 300 m³ compost and using the measured bulk density of 391 kg m^{-3} compost; Table 7.8). This is the minimum price that *Het Bolhuis* should ask to the compost users to not make economic losses. In case part of the compost is for own use, the cost for paying a contractor to spread this compost on his own fields needs to be added, amounting the total cost to **€ 8.2 t^{-1}** compost (Table 7.8).

In **alternative scenario C**, when composting annually 300 m³ FYM, 300 m³ pre-composted hay from *Natuurpunt* is needed, which is transported for free by *Natuurpunt* to the farm. *Natuurpunt* proposed a cost price for the pre-composted hay of one third of the production cost (€ 5.6 m^{-3}). As such, *Het Bolhuis* paid € 2 m^{-3} pre-composted hay to *Natuurpunt*. Similarly to scenario B, the farmer buys the compost covers and thermometer, and pays the transport from the manure to the field. Turning costs and using the compost turner is for free, in exchange for valorizing the hay from *Natuurpunt*. This gives a total cost of € 4.2 m^{-3} compost or **€ 12.6 t^{-1}** compost (when producing 300 m³ compost and using the measured bulk density of 330 kg m^{-3} compost; Table 7.8). This is the minimum price that *Het Bolhuis* should ask to the compost users to not make economic losses. In case part of the compost is for own use, the cost for paying a contractor to spread this compost on his own fields needs to be added, amounting the total cost to **€ 15.3 t^{-1}** compost (Table 7.8).

Table 7.8 Cost calculation for the business as usual (BAU) and alternative scenario for case study 3 (€ $tonne^{-1}$ end product, left column), based on the amount of end product (FM) produced in this particular case study (t $year^{-1}$, right column). OM = organic matter, FM = fresh matter.

	BAU scenario		Scenario A		Scenario B		Scenario C	
	Cost (€ t^{-1})	Amount (t y^{-1})	Cost (€ t^{-1})	Amount (t y^{-1})	Cost (€ t^{-1})	Amount (t y^{-1})	Cost (€ t^{-1})	Amount (t y^{-1})
Production	0	300	7.7	150	5.5	300	12.6	300
Spread on own fields	2.7	300	2.7	150	2.7	300	2.7	300
Total cost on FM base*	0 or 2.7	300	10.4	150	8.2	300	15.3	300
Total cost on OM base*	0 or 18.0	45	71.6	22	40.4	61	61.7	74

* if spread on own fields

7.3.3. Regulation

First, a general overview of the regulation concerning on-farm composting is given. Hereafter, for each of the case studies, the specific regulation, theoretically applicable if the alternative scenario(s) would be implemented, is summarized.

7.3.3.1. General overview

The European, Federal and Flemish legislation are covering manure storing and handling, composting, trading and applying compost (with or without manure). There are five types of regulations: (1) regulation about installing a compost facility and urban development permit, (2) Flemish Environmental Permitting Regulations, (3) other Flemish (environmental) regulations (waste, manure, soil, etc.), (4) Federal regulation for trading and (5) the European regulation. In general, the use of manure determines if composting is classified as 'waste treatment' or 'manure treatment'. Below, we give an overview of the most relevant regulations regarding (on-farm) composting.

Table 7.9 Standard costs and data for calculating total costs of compost use from an industrial compost facility and production and use of on-farm compost.

	Unit	Mean	Min	Max	Remark	*Source*
Use of compost from compost facility						
Variable						
Purchase of compost	€ t^{-1}		2	25	Dependent on facility, quality, sieving, additives, …	*Online data search*
Loader capacity	m^3		10	45	Must be < 30 t	*Data case study 1*
Transport of compost by contractor	€ h^{-1}		48.5	52.5	Dependent on loader capacity	*Data case study 1*
Spread of compost by contractor	€ h^{-1}		48.5	52.5	Dependent on loader capacity	*Data case study 1*
Spread of compost by contractor	m^3 h^{-1}	30			Dependent on loader capacity and dose per ha	*Data case study 1*
Production and use of on-farm compost						
Fixed						
Pile cover	€ m^{-2}	2.18			50 m length	*Online data search*
Temperature sensor	€		22	275	Dependent on type, new/second hand	*Online data search*
CO_2 sensor	€		420	490	Dependent on type, new/second hand	*Online data search*
Compost turner	€		10000	35000	Dependent on type, new/second hand	*Online data search*
Variable						
Shredding wood chips	m^3 h^{-1}		10	30	Wood in wood edges, spreaded (10m^3) or wood are already on a heap (30m^3)	*Stakeholder consult*
Contractor for shredding wood	€ h^{-1}	150				*Stakeholder consult*
Set-up of piles with telescopic handler	m^3 h^{-1}	30				*Data case study 2 and 3*
Hiring of compost turner	€ h^{-1}	55			Dependent on agreement between partners	*Data case study 2 and 3*
Turning pile	h $pile^{-1}$	0.5			Including coupling compost turner to tractor	*Data case study 2 and 3*
Turning pile	frequency per compost event		1	8	Dependent on feedstock and process	*Data case study 2 and 3*
Monitoring pile	h $pile^{-1}$	0.3				*Data case study 2 and 3*
Monitoring pile	frequency per week	2				*Data case study 2 and 3*
Composting period	weeks		8	12	Dependent on feedstock and process	*Data case study 2 and 3*
Fuel consumption	l h^{-1}		10	25		*Online data search*
Fuel price	€ l^{-1}	0.6			Depent on moment	*Online data search*
Spread of compost by contractor	€ h^{-1}		48.5	52.5	Dependent on loader capacity	*Data case study 1*
Spread of compost by contractor	m^3 h^{-1}	30			Dependent on loader capacity and dose per ha	*Data case study 1*
Fresh bulk densities (mass <-> volume)					Dependent on moisture content	
Compost	kg m^{-3}		500	800		*Data obtained from PhD research*
Grass clippings	kg m^{-3}		40	300		*Data obtained from PhD research*
Wood shavings	kg m^{-3}		200	400		*Data obtained from PhD research*
Cattle farmyard manure	kg m^{-3}		300	700		*Data obtained from PhD research*

(1) Installing and urban development regulation

When installing a compost facility (total volume > 10 m³), an urban development permit from the community is required (Flemish Decree on Spatial Planning, Belgian Official Gazette, 15 March 1997). When composting animal manure, you need to take into account the Circular RO/2006/0137 for installing plants for manure treatment and digestion (e.g., maximum input of 60 000 t $year^{-1}$). Further, the recognition of composting plants using animal manure are also related to the European regulation 1069/2009 (see below, number (5)).

(2) Flemish Environmental Permitting Regulations: VLAREM

The Flemish Environmental Permitting Regulations are important for storage of feedstock materials and compost, and the composting process itself. VLAREM I describes the procedures and classifications of the environmental license permits, while VLAREM II describes the terms that the licensed facilities need to fulfill. VLAREM I differentiates three classes of 'hindering' facilities: class 1 (most hindering) and class 2 (less hindering) need to have an environmental license (including environmental terms), while facilities in class 3 only have a reporting requirement. The environmental class of the compost facility is dependent on the category, listed in Appendix 1 of VLAREM I, which is dependent on the origin of the composted materials.

In **category 2**, when composting only **plant-based** materials, especially the storage or processing capacity of the composting plant determines the class. Below a volume of 25 m³: class 3 (reporting requirement), between 25-2000 m³: environmental license class 2 and > 2000 m³: environmental license class 3. However, for on-farm composting, with *feedstock materials from the own farm* and *using the compost on the own fields*, no environmental license is needed. However, most of the farms also compost feedstock materials from other parties, as they do not have the appropriate feedstock materials to have an optimal feedstock mixture available on their own farm (e.g., most farmers have a shortage in C-rich feedstock materials). This means that on-farm composting of plant-based residues often requires a class 2 environmental license.

Category 9 applies to manure treatment on animal farms, only when treating (composting) **own manure** and other plant-based residues from the own farm. Dependent on the livestock densities and region, a class 1 or 2 environmental license is required.

Category 28 applies to **other manure treatment** plants not under category 9. The plants are divided based on the annual capacity of feedstock treated. An environmental license class 2 is sufficient between 2-1000 t, above 1000 t a class 1 is required.

VLAREM II describes the terms, related to the environmental class (preventing odor, emission limits, management and capturing of waste water, etc.). More specifically for composting, the installation of a concrete pad with a system to capture and store run-off waste water is an expensive investment for a farmer.

An overview of the VLAREM regulations was already given in Chapter 2, Table 2.2.

(3) Other Flemish regulations concerning waste and manure

The Flemish regulation concerning the sustainable management of materials and waste (VLAREMA) states that only compost from a licensed compost facility, that fulfills the quality requirements of VLAREMA, can leave the waste statute. The Flemish Compost Organization (Vlaco vzw) or another recognized certification organization controls and grants the certificate, which of course is an administrative cost for the farmer.

For **compost application and transport**, the Flemish Manure Decree is applicable. Manure-based compost belongs to the category 'animal manure'. The maximum dose animal manure that can be applied is 170 kg N ha^{-1} . Plant-based compost belongs to 'other fertilizers' and can be applied along with animal manure and mineral fertilizers, as long the N and P fertilizer limits are not exceeded. For certified green waste compost or vegetable, fruit and garden waste compost, only 50% of the P_2O_5 content needs to be taken into account. Only 15% of the N input with certified green waste compost or vegetable, fruit and garden waste compost is considered to be active, while for farm compost, the 15% active N rule is not applicable. For farm compost and FYM, the 50% P_2O_5 rule is only applicable on P_2O_5 class I and II parcels. Compost (> 500 kg) must be transported from the compost producing company to the user by a certified manure transporter or certified sender.

The **storage of solid manure** (farmyard manure, champost, solid fraction after separating slurry and manure with a DM content ≥ 20%) on the field is forbidden from the 15th of November until the 15th of January, and is limited to maximum two months outside that period (see Chapter 5). This implies the need for on-farm storage under strict conditions, including the installation of a concrete, impermeable pad with three concrete walls, which makes composting impossible.

(4) Federal regulation for trading compost

The Royal Decree of 28 January 2013 on the marketing and the use of fertilizers, soil improvers and growing media (not valid for manure) determines that a fertilizer needs to be on the list in Appendix I to be traded in Belgium. End products from composting are not on the list, therefore an exemption must be asked to the federal public service 'Health, Food chain safety and Environment'. There are standard federal norms for green waste compost, vegetable, fruit and garden waste compost and vegetable, fruit and garden waste compost from composting after anaerobic digesting. For manure and manure-based compost no such an exemption is required (natural product from the farm).

Further, when trading manure-based products, the European regulation 1069/2009 is applicable (see also point (5) below). When fulfilling the requirements in the regulation, digestate and animal-based compost can be transported to other European countries as (resource for) soil improvers.

(5) European regulation

The European regulation 1069/2009 states the health rules for non-human consumption of animal byproducts, including collecting, transporting, storage, handling, processing and using or exporting. Manure is conform the regulation a 'category 2' material. The regulation includes terms for the processing installations as for the end products: e.g., the installations must be recognized by the National Manure Bank, and the end products must by sanitized by heating at 70°C for minimum 1 hour.

7.3.3.2. Alternative scenario case study 1

- Co-composting green waste (nursery residues) from the own farm with manure from a third party belongs to Category 28 from Vlarem I (= manure treatment). The production is below 1000 t, hence, an environmental class 2 is needed.
- VLAREMA: a controlling organization needs to check the quality requirements.
- Urban development permit for the compost installation (including a concrete pad) is necessary.
- Because the compost is used on the own fields (no trade or transport), only the Manure Decree is applicable for compost application. All the active N and total P_2O_5 content originating from the manure needs to be taken into account for calculating the fertilization dose.

7.3.3.3. Alternative scenario case study 2

- Co-composting own farmyard manure with C-rich materials from a third party (nature area) probably belongs to Category 28 from VLAREM I.
- VLAREMA: a controlling organization needs to check the quality requirements.
- Urban development permit for the compost installation (including a concrete pad) is necessary.
- Because the compost is used on the own fields (no trade or transport), only the Manure Decree is applicable for compost application. All the active N and total P_2O_5 content need to be taken into account for calculating the fertilization dose. However, it is unclear how the nutrients from extern plant-based C-rich byproducts should be taken into account.

7.3.3.4. Alternative scenario case study 3

Scenario A

- Composting own farmyard manure and trading the compost, probably belongs to Category 28 from VLAREM I and needs an environmental class 2.
- Compost is not for own use only, a controlling organization needs to check the quality requirements.
- To trade the compost, no exemption from the federal public service is necessary (natural product from the farm).
- To transport the composted manure to a farm in the same or an adjacent municipality, no certified transporter or sender is necessary, the farmers can make use of the 'neighbor arrangements' and can use the analyses from the case study. Otherwise, a certified transporter is necessary.
- All the active N and total P_2O_5 content originating from the manure needs to be taken into account for calculating the fertilization dose.

Scenario B

- Similar to case study 2, except that the compost is also transported (and hence considered traded) to other farms. Because the used grass clippings are originating from parcels with a parcel user agreement, they could be considered as proprietary feedstock, and then a category 9 would be sufficient. However, because the compost is also transported to other farms, probably a category 28 is applicable.

- The end product in our view can still be considered a natural product from the farm, as the used grass clippings are originating from parcels with a parcel user agreement. Hence, we assume no exemption from the federal public service is needed.
- Compost is not for own use only, a controlling organization needs to check the quality requirements.
- To transport the composted manure to a farm in the same or an adjacent municipality, no certified transporter or sender is necessary, the farmers can make use of the 'neighbor arrangements' and can use the analyses from the case study. Otherwise, a certified transporter is necessary.

Scenario C

- Similar to scenario B.
- Probably an exemption from the federal public service is needed to use the pre-composted residues obtained from *Natuurpunt* as feedstock material for composting.
- If a redemption is needed, a certified transporter or sender should be used to transport the compost to the other farmers.

7.4. Discussion

7.4.1. Compost quality

Generally, besides soil improving, manure-based farm compost has a higher fertilizer value (contains more N, P and K) and has a lower C/N and C/P ratio compared to green waste compost. This could be considered as a weakness of farm compost, since less of the product can be applied following the fertilization standards and, consequently, less OM can be applied. However, a comparison of the stability of the OM should be made. Moreover, the P in farm compost that originates from own plant material should not be considered, applying the fertilization standard for P. However, it is not clear how to account for N from farm compost, applying the fertilization standard for active N. Further, the final nutrient content of the compost is related to the nutrient content of the initial feedstock materials. For example, the three composts in case study 3 had a higher N/P ratio than green waste compost, related to the higher N/P ratio of the initial farmyard manure. As such, using feedstock materials with a high C/P, C/N or N/P ratio (e.g., fresh wood chips) results in a compost with a high C/P, C/N, or N/P ratio. The three different case studies showed that the cooperation

between farmers and third parties resulted in an end product of higher quality when compared to applying/valorizing only the own byproducts (wood chips in case study 1 and FYM in case studies 2 and 3). As it was a first experience with composting for the majority of the partners, it can be assumed that both the composting process, and the quality of the end products will further improve by experience and learning. For example, in case studies 1 and 2, the OM content of the composts can be increased by minimizing the soil particle load of the feedstock materials. Less soil would be shoveled up by, e.g., using an absorption layer of hay or sowing grass early enough on the place of composting, or by composting on a concrete pad. In case study 2, next time, slurry should be added in a more stepwise way, to prevent oxygen shortage, nutrient losses and slowing down of the composting process by adding too much moisture at once.

7.4.2. Socio-economic aspects

The organization of on-farm compost production depends on a number of socio-economic and contextual factors. The three action research cases illustrated five general factors that determine the choice of production set-up: (1) the available biomass, (2) the local opportunities, (3) the local compost requirements, (4) the agreements made between partners, and (5) the facilities and capabilities of the partners.

(1) *Available biomass*. A farmer needs to compost with the biomass that is available, on his or her own farm, or in the neighborhood. This latter can be from adjacent nature areas (sod cuttings or chopped material from heathland management, reed from bank management, litter from forest conversion, etc.) or from other farmers (spent growing media from horticulture, farmyard manure, etc.). For instance, in case study 1, *SYLVA* only has woody residues available on the farm. They use available FYM from a neighboring farmer to co-compost with the woody residues. However, it should be economically feasible for either the supplier of the biomass or for the farmer to transport it to the farm to compost it. For example, in case studies 2 and 3, *Natuurpunt* needs to consider what the best option is to valorize their byproducts, taking into account the transport distance and cost, gate-fee at an industrial composting plant, pre-composting the biomass or not, asking a price for the biomass or giving it away for free, etc. In case study 2, if the transporting distance of the grass clippings from the nature area to the farm would be over 20 km, it would be financially favorable for *Natuurpunt* to bring the grass clippings to an industrial composting plant if that

distance is under 10 km, instead of bringing it to the farm and helping by the set-up of the compost pile.

(2) *Local opportunities*. A farmer is dependent on opportunities at the moment for biomass availability and cooperation possibilities for e.g., composting equipment. For example, in case studies 2 and 3, the farmer is not able to compost himself, but he can hire a compost turner from *Natuurpunt*[4]. In Flanders, also the Research Centre for Vegetable Cop Production Kruishoutem[5] and *vzw 't Boerenlandschap*[6] are letting compost turners. Another option is a service provider in the neighborhood who executes the composting process, as was the case in case study 1 where the composting process was executed by a private company producing compost. However, this could also be done by a contractor, a cooperative contractor such as *Agro|aanneming*[7] or Agrobeheercentrum Eco²[8] in Flanders.

(3) *Local compost requirements*. The requirements for compost quality can influence the choice of the used biomass. For example, in case study 3, when there is need for compost in organic farming for improving the SOM content, a farmer can choose to compost with byproducts from nature areas that are rich in C and low in nutrient content. If there is more need for nutrient-rich compost, more manure can be used as feedstock material.

(4) *Agreements*. The type of cooperation and possibilities for compost production is dependent on the agreements and trust between the partners. An example of the economic impact of variable agreements between partners is that in case study 3, *Natuurpunt* is turning the piles for free in exchange for valorizing their nature residues, while in case study 2, a cost of € 75 per week is charged to the farmer. Further, in the BAU scenario of case study 1, the farmyard manure from the neighbor is transported and spread for free on the fields of *SYLVA*. However, when *SYLVA* has to pay for this, the BAU could become more expensive.

[4] https://www.natuurpunt.be

[5] http://www.pcgroenteteelt.be/nl-nl/

[6] http://www.boerenlandschap.be/

[7] http://www.agroaanneming.be/over-agroaanneming

[8] http://www.agrobeheercentrum.be/

(5) *Facilities and capabilities of the partners.* Dependent on the financial, time and technical capabilities of the partners, certain practices and collaborations are excluded or necessary. For example, when there is no time to monitor the composting process, the farmer needs to outsource this to a service provider, as was the case in case study 1.

Consequently, the production costs for on-farm composting are also depending on the above mentioned influencing factors. As we did not account for these factors, our results are rather explorative, hence, no unequivocal conclusions can be drawn from the comparison of the production costs over the different composting scenarios. Therefore, we want to highlight that the calculation of the costs are only a first step for economically comparing the different valorization options. To gain more insights into different scenarios for on-farm composting, more accurate estimates of both the composting and costs variables, and the environmental benefits are required, allowing a more detailed economic study. Further, it must be noted that in none of the on-farm composting scenarios, we took into account the costs for a concrete pad to compost on, provided with a system to capture and store run-off waste water, according to the requirements when using external feedstock (further discussed in section 7.4.3 below). This would be an extra cost of € 50 m^{-2} concrete, which is a huge investment for an individual farmer. Nevertheless, a number of general trends seem to emerge from our calculations.

- On-farm production seems generally more expensive (between € 5.5 t^{-1} in scenario B, Table 7.9; and € 26 t^{-1} in case study 2, when buying the compost equipment) than buying compost from an industrial composting facility (€ 2.5 t^{-1} in case study 1, Table 7.7). However, the possible added value from a better soil improving product and less environmental impact by recycling farm residues, is difficult to take into account in a cost analysis.
- The stability of the applied OM is different for each product. Therefore, in all cases it is more expensive to apply e.g., farm compost compared to uncomposted FYM, but more stable OM will be added and hence the SOC content will increase more with the amendment of compost compared to CFM.
- The cases seem to show that cooperation between different partners decreases the costs for compost production. For example, the production cost is € 8 t^{-1} less expensive when cooperating with *Natuurpunt* for biomass and compost equipment compared to composting by one farmer in case study 2 (Table 7.8), and € 2 t^{-1} less

expensive in case study 3 (Table 7.9). The success of the collaboration of course depends on trust and good agreements. Unfortunately, the regulations for exchanging biomass are not stimulating collaboration (discussed below in section 7.4.3).

- Paying a service contractor to execute the composting process will only be economically advantageous when enough biomass is composted, as the transport costs to set-up and monitor the pile stay the same (case study 1). As such the cost per tonne compost will decrease when more compost is produced.
- In most cases, buying the composting equipment is more expensive than hiring it. However, at a certain production volume, buying composting equipment will be the most economical solution, as the depreciation cost (such as composting equipment) decreases per additional tonne of compost produced. This was demonstrated by the lower production cost in case study 1 (€ 8.1 t^{-1} for 800 t compost) compared to case study 2 (€ 26 t^{-1} for 120 t compost). However, it will be difficult to reach high production volumes due to the small-scale farming in the considered cases. Potential solutions to make it economically more feasible to buy the composting equipment could be to let it to other farmers and/or to sell the compost, although other regulations are applicable in the latter scenario (see section 7.3.3.1).

In line with our conclusions in Chapter 2, we could observe in these cases that, currently, the valorization market for byproducts generally is driven by (1) the challenge or necessity to handle "waste" (organic-biological byproducts which, for most people, seem to have no immediate value anymore) and (2) the costs related to this processing. The ultimate value of the end product seems to be of secondary importance in deciding which processing and destination to choose. The nomenclature for compost products is indicative for these drivers: we use the terms "green waste compost" or "vegetable, fruit and garden waste compost" and make no reference to the value of the end product (e.g. "compost for horticulture").

As such, e.g. the gate-fee, demanded by larger-scale composting plants strongly influences both compost use and "waste" or feedstock destination: this industrially produced compost is cheaper than on-farm production of compost, because a significant part of the cost is covered by this (often quite high) gate-fee. On the other hand, for producers of byproducts (e.g., nature conservationists or animal farmers), this gate-fee is an obstacle and its cost (together with the transport cost) will be balanced against costs for bringing these byproducts to a local farm and, given the reduced costs, even own investment in composting infrastructure or material and aid to farmers willing to further process these byproducts.

Despite this general trend, we could observe that in some cases the need to invest in soil quality has become so high and has an immediate impact on productivity (e.g., for tree nurseries), or the fertilization restrictions have become so critically (e.g., in organic farming), that people are willing to make extra costs and efforts. We could prove that, in our case-studies, there is a high demand for qualitative on-farm compost, in combination with the valorization of biomass byproducts, despite the higher production cost and regulatory restrictions, because in all cases the farmers kept on investing in on-farm composting, even without our support.

Of course, these cases are probably quite exceptional and these sectors and farm types only cover a small share of the overall Flemish agriculture. To make a transition from a waste/cost driven market to a market where the value of the end product is of major importance during biomass valorization, some kind of financial stimulus, policy incentives and/or better dissemination will be necessary. To motivate this however, more accurate and integrated estimates of the environmental benefits of both composting and compost use are required. Further, society must be convinced that biomass byproducts are not "waste products", but valuable resources.

7.4.3. Legislative aspects

The limitations as a consequence of the complex and often not-stimulating regulation for on-farm composting, as was already demonstrated by the stakeholder consult (Viaene et al., 2016b, Chapter 2), were also clearly observed in all three cases in our action research. Especially, restrictions in the use of extern biomass and the related environmental license are discouraging farmers to compost. The latter include the requirement for a concrete pad to compost on and inspections by quality control organizations. Even when using biomass from parcels for which the farmer has a parcel user license (as in case study 3), the biomass is normally not considered as proprietary. Consequently, an environmental license is needed when using it as a feedstock for composting. Nevertheless, when different feedstock materials are combined and thereby the feedstock mixture is optimized, the composting process and thus the compost quality will improve and nutrient losses will decrease, as was also demonstrated in the former chapters of this thesis. We proved in case study 3 that co-composting FYM with grass clippings from nature areas (scenarios B and C) significantly reduced the N losses to soil compared to only composting FYM (scenario A; Viaene et al 2016a). Moreover, residues from nature management are generally free from pollutants such as pesticides, herbicides and garbage. Therefore, we believe that extending the definition of

on-farm composting, now restricted to one farmer who composts own biomass and applies compost on own fields, would stimulate on-farm composting. More specifically, farmers, composting relatively small amounts on occasional base and for own use, should be excluded from the environmental license requirement when using extern biomass from nature areas. Of course they should take some precautions, e.g., cover the piles with a geotextile, compost on different locations, monitor the piles, compost on an organic layer (e.g., straw), etc.

Besides using extern biomass, also the cooperation between different partners resulted in a better and economically more feasible composting process. However, to date, trading compost (even for free) is not possible without environmental license, quality control and exemption from the federal public service 'Health, Food chain safety and Environment'. Therefore, we think that the cooperation within a small and clearly delineated group of nearby farms to compost their byproducts could be considered as on-farm composting and the compost could be considered as a natural product from the farm. In that way, locally available byproducts, too small in volume for other valorizations, are valorized on a local scale and the compost can be exchanged between farms without extra financial investments and administrative burdens. Of course, restrictions on the type of biomass, amount of compost production, transport distance, etc. should be defined.

7.5. Conclusion

To gain more insight in the feasibility (in terms of compost quality, costs and regulation) of collaborative production forms of on-farm composting, three action research case studies were performed on different locations in Flanders, in collaboration with stakeholders and relevant policy advisors and implementers. The cases indicated that cooperation between different partners for the production of on-farm compost resulted in a better and economically more feasible composting process compared to composting by an individual farmer. However, to make a transition from the current waste/cost driven valorization market for byproducts and compost production, to a market where the value of the end product is of major importance, better estimates of environmental impact and some kind of financial stimulus, policy incentives and/or better dissemination will be necessary.

CHAPTER 8

General discussion and conclusions

Local on-farm composting fits within the view of a sustainable bioeconomy and agricultural systems, in which sustainable soil management and other agro-ecological practices have an important role to play, because (1) organic-biological byproducts are valorized, (2) biomass from prior biomass processing (cascading use of biomass) or biomass unsuitable for other processes can be valorized, (3) its production is less dependent on non-renewable resources, (4) nutrient and material cycles are closed on a local scale, and (5) compost application contributes to soil quality and fertility. However, despite these advantages, both composting and compost application are not common practices in (Flemish) agriculture. Therefore, the objective of this thesis was to **(1)** identify the challenges and hindering factors to on-farm composting and the application of compost in agriculture. Those barriers were used to further refine the aims of the thesis: **(2)** increasing insight in the processes, environmental impact and application value of on-farm composting with locally available organic residues (technical objectives) and **(3)** providing tools for farmers, policymakers and other stakeholders, ultimately stimulating feasible, efficient and sustainable on-farm composting and use of high-quality compost in Flemish agriculture (socio-economic objectives).

In order to address both technical and socio-economic challenges of on-farm composting, and to increase the chance of adoption of on-farm composting, an innovative, transdisciplinary research approach was used, in which continuous interaction between the different stakeholders was of major importance.

In this concluding chapter, the results from Chapters 2 to 7 are synthesized and some general reflections are made. In sections 8.1 and 8.2, feedback is given to, respectively, the technical and socio-economic challenges of on-farm composting as pointed out in Chapter 1, Figure 1.1, being:

(1) The optimization of composting techniques for ‘difficult’, underutilized and/or fast degradable farm residues
(2) The shortage of C-rich feedstock materials
(3) The risk for N losses
(4) The quality assessment of composts

(5) Legislation, logistics, financial and time investments

Next, in section 8.3, the lessons learned from the innovative GENESYS approach are summarized. Finally, in section 8.4, suggestions for future research are proposed.

8.1. Technical challenges and hindering factors

8.1.1. Optimization of composting techniques for 'difficult', underutilized and/or fast degradable farm residues

During this thesis, we searched for valorization options for three 'difficult', underutilized and/or fast degradable farm residues: N-rich vegetable crop residues (Chapters 3 and 4), cattle farmyard manure (Chapters 5 and 7) and solid fraction of cattle slurry (Chapter 6). Considering the vegetable crop residues, leek residues are a good example of a fraction already collected during standard harvesting operations, whereas for many other vegetable residues extra efforts should be made to collect them (Agneessens et al., 2014). We compared ensiling, composting and anaerobic digestion (only for the crop residues), and concluded that the three options can be considered as good strategies to process those residues. Appropriate reapplication of silages, composts and digestates to soil contributes to soil quality and closes nutrient cycles on a local scale. Therefore, the on-farm facilities and different soil effects of processed vegetable crop residues (discussed further in this paragraph) should be taken into account when selecting the most appropriate valorization option and application in time and space.

It is well known that an optimal combination of 'green' (N-rich) and 'brown' (C-rich) feedstock materials is a prerequisite to ensure a good composting process, i.e., sufficient oxygen, water, N and C are necessary to stimulate aerobic microorganisms and OM degradation and stabilization. This was clearly confirmed by the experiments conducted during this study. For example, in Chapter 6, we showed that pure composting of solid fraction resulted in oxygen shortage, and thus a suboptimal composting process. In case study 2 from Chapter 7, the application of too much N and moisture at once slowed down the composting process and increased N losses. A possibility is to add leek residues sequentially to the compost pile, which is also according to regular farm practices (Chapter 3). Adding C-rich bulking agents to N-rich residues resulted in more available C and structure, a decrease in the moisture content and an increase in the oxygen supply (Chapters

5 and 6). Further, in Chapter 5, we proved that by optimizing the feedstock mixture, higher pile temperatures were achieved, leading to higher OM degradation, sanitation and stability of the end product. Moreover, a combination of fine and coarse bulking agents resulted in the best composting process (Chapter 4). **Composting** resulted in a drier and **more stable and sanitized** end product with less, but **more recalcitrant OM**, less mineral N and a **higher N/P and C/P ratio** compared to silages, which make composts suitable for export.

Composting requires appropriate equipment, time and knowledge. Hence, it is important to also consider options which require less labor, time and machinery. For crop residues (Chapter 3) and solid fraction (Chapter 6), co-ensiling with drier and more C-rich bulking agents is a potential conservation option. In Chapters 3, 5 and 6 we showed that by **co-ensiling**, OM and nutrients are more conserved during storage compared to composting, resulting in a more **biodegradable product with more plant-available nutrients** compared to compost. Therefore silages are preferred to local use. However, direct application of vegetable based silages caused temporary N immobilization (Chapter 3). Therefore, a large enough time interval between field incorporation of the silages and crop sowing date should be considered. In contrast, application of silages after co-ensiling solid fraction with a mixture of straw and grass did not lead to N immobilization; we even noticed a twice as high N mineralization compared to compost application (Chapter 6). This could be explained by a higher C availability (as indicated by a higher biodegradation potential and OUR) of the silages based on crop residues compared to the silages based on solid fraction.

After ensiling, the silages can be used as feedstock for anaerobic digestion to produce **bioenergy**, after which the remaining **digestate** can be reapplied to the soil. This was tested for crop residues (Chapter 3). Compared to compost and silages, digestate contained more plant-available N and therefore acts as a **short-term fertilizer**.

8.1.2. Shortage of C-rich feedstock materials

N-rich feedstock materials should be mixed with more coarse, C-rich bulking agents to have an optimal composting process (Barrington et al., 2002). In Chapter 2, we identified that commonly used bulking agents such as wood chips are not always available for the farmer and are, in that case, an expensive feedstock, hindering farmers to (produce good quality) compost. In the search for alternative C-rich bulking agents to mix with more N-rich farm residues, a farmer could use byproducts from nature management or other nearby farms (e.g. from greenhouse cultivation or tree nurseries). We tested chopped heath biomass and spent

growth media from strawberry and tomato cultivation as examples for bulking agents to replace wood chips in co-composting leek residues and concluded that all three byproducts have potential (Chapter 4). When using chopped heath biomass, we could produce a compost with a low nutrient content and high C/P ratio (232), valuable to increase the soil organic matter (SOM) content in P-rich soils and to replace peat in growth media. When using spent growth media, it was required to add a coarser bulking agent to prevent oxygen shortage. Further, using spent growth media resulted in lower OM degradation compared to chopped heath biomass, and a nutrient-rich compost. The results of Chapters 3, 4, 5 and 6 clearly showed that the variability in compost composition is mainly related to the initial feedstock materials; therefore, it is important to start from a (bio)chemical characterization of the used feedstock materials in order to predict the composting process and compost quality.

The selection of the most appropriate bulking agents depends also from the context-specific soil and crop requirements and the local availability (to reduce transport costs) of bulking agents. Dependent on local availability, also other residues than the ones tested during this study can be used. We give an overview of the amounts and seasonal availability of other, currently under-valorized, C-rich byproducts in Flanders that can be used as bulking agent (Table 8.1, non-exhaustive list). This table shows that there is high opportunity (in terms of available quantity) for the use of grass clippings from roadside management and woody biomass from gardens, parks, tree rows, etc. However, those residues are usually processed in large-scale composting facilities, which seems the most appropriate valorization option taking into account the potential presence of garbage and the high volumes available on a short time period. For on-farm composting we see more potential in the use of woody biomass from small landscape elements, residues from nature areas, tree nurseries and horticulture, as they are not contaminated with garbage, year-round available and currently under-valorized. As such, competition with large-scale composting facilities is avoided as well. However, estimating the quantities remains difficult, as these byproducts have a fragmented ownership and geographic availability. As such, the opportunity to use them depends on local availabilities and agreements between parties. Furthermore, it is important to know that, when using external biomass for on-farm composting, an environmental license (which brings, e.g., the financial investments of a concrete pad to compost on) is generally required. This legislative issue is further discussed in section 8.3.

Table 8.1 Overview of the amounts and seasonal availability of currently under-valorized, C-rich byproducts in Flanders that can be used as bulking agent in composting. FM = fresh material.

Byproduct	Amount t or m³ FM ha^{-1}	Area ha	Total amount t or m³ FM year^{-1}	Availability	Source
Spent tomato substrate (data from 1 grower)	35 t	9.2	322 t	December	[1]
Spent strawberry substrate	35 t	293	10255 t	March-December	[1] and [2]
Woody residues from berry cultivation	2 t	301	456 t	Year-round	[1]
Woody residues from tree nurseries	6 - 9 t	4638	34785 t	Year-round	[1] and [3]
Litter from forest conversion	249 t	6	1493 t	Winter	[4]
Sod cuttings from heath management	261 t	56	14621 t	Half August-February	[4]
Chopped heath from heath management	87 t	74	6438 t	Half August-February	[4]
Grass clippings from grassland management	7 t	6387	42178 t	Peaks in June and September	[5]
Grass clippings from roadside management	6 t	23501	144063 t	Peaks in June and September	[5]
Woody biomass from small landscape elements, gardens, parks, tree rows, etc. (informal, private market)	3.5 m³	80000	430000 m³	Year-round	[6]

[1] Stakeholder consult, [2] Platteau et al. (2014), [3] Meeusen et al. (1998), [4] Gybels et al. (2013), [5] Delief and De Vocht (2012), [6] Vandekerckhove et al. (2014)

8.1.3. Risk for N losses

Nitrate losses by leaching, related to the fast decomposition of fresh N-rich crop residues left on the field or leachate losses from stored manure piles, may have a negative impact on surface and groundwater quality. However, this can be prevented or minimized by treating and stabilizing N-rich residues in a controlled way (Rotz, 2004). In Chapter 3, it was shown that composting, ensiling or anaerobic digesting of fresh vegetable crop residues after harvest allow to process and/or store crop residues over winter. The incubation experiments in Chapter 3 demonstrated that reapplication of composts, silages and digestates to the soil did not lead to significant N mineralization. When storing farmyard manure on the field, N losses to the soil can be reduced by mixing C-rich bulking agents to have an optimal feedstock mixture for composting and covering the pile with a geotextile (Chapter 5). However, this is quite a challenge, and the outcome of a particular treatment (composting, ensiling, covering with a geotextile) was strongly dependent on the initial product characteristics and storage conditions, as clearly observed in Chapters 3 and 5. Nonetheless, in Chapter 5, we showed that N leachate losses during storage of farmyard manure are limited (max. 4.2% of the initial manure N content). Further, they could be mitigated, for example, by annually alternating the location of the manure piles, with sufficient distance from vulnerable zones such as watercourses. These results allow policy makers to reconsider the legislation for manure storage in Flanders. Currently, it is prohibited to store manure on the field between 15/11 and 15/01 and restricted to maximum two months outside this period to avoid N leaching. Compared to the surrounding regions, there are clear differences in legislation. In the Walloon region of Belgium, farmyard manure with sufficient straw content (depending on specific stable conditions) and compost with a minimal DM content of 35%

can be stored on the field for a maximum of 10 months, under certain conditions such as a minimal distance of 20 m from watercourses and not on a slope of more than 10%[9]. In the Netherlands, farmyard manure can be stored on the field for two weeks without precautions. If the storage period is longer than two months but less than 6 months, it is sufficient to store farmyard manure 5 m from a vulnerable zone, under a cover and on an absorbing layer of a minimum of 15 cm and 25% organic material (e.g. straw). Only when the storage period is longer than 6 months, it is necessary to put the farmyard manure on a concrete floor including the collection of rainwater and leachate[10].

Leachate losses (Chapter 5) were low compared to reported volatile N losses during composting (11-51% of the initial N content; Martins and Dewes, 1992). Therefore, it is important to take into account the N losses, both liquid and volatile, over the whole management chain: from stable (in the case of manure) or field (in the case of crop residues), over storage and processing, during spreading and after field application (uptake by the succeeding crops) to consider the best management strategy. In addition, the losses should be carefully weighed against the environmental impact of other alternatives. For example, prohibiting field storage of manure will prevent leachate losses to the soil during storage. However, unmanaged storage on a concrete floor may induce volatile losses (Kolenbrander and De la Lande Cremer, 1967). Covering with a plastic (anaerobic storage) will conserve more N during storage compared to composting, however, higher losses after application are expected, if not taken extra measures (e.g., manure incorporation; Shah et al., 2013).

8.1.4. Quality assessment of composts

8.1.4.1. Variation in compost parameters

The results from Chapter 2 indicated that the lack of information on compost quality and composition is hindering farmers to apply compost. Generally, farmers apply compost for maintaining or improving overall soil health and quality, and more specifically the SOM content. Therefore, it is important to apply compost with a high content of stabilized OM. However, also a considerable amount of nutrients are added when applying compost, while the Flemish manure legislation limits the amounts of N and P that can be applied by organic and mineral fertilizers, depending on the crop and soil type (Tits et al., 2014). This implies

[9] https://emis.vito.be/sites/emis.vito.be/files/legislation/1332/2014/sb120914-2.pdf

[10] http://veldleeuwerik.nl/opletten-bij-opslaan-vaste-mest/

that when applying organic soil improvers for improving SOM content, products with a high C/N and C/P ratio are preferred.

To have a general idea about the composition of different compost types, we summarized all compost parameters from the end products of the compost experiments conducted during this PhD research (Figure 8.1). The composts were grouped based on the used feedstock materials, i.e., cattle farmyard manure (n = 29), solid fraction of cattle slurry (n = 16) (together called *manure based composts*) and vegetable crop residues (*vegetable based composts*, n = 32). Boxplots were plotted to compare the distribution of the parameter between and within groups. It can be clearly seen that the median OM content is not strongly different between the three compost types. However, in Chapters 2 and 7 it was shown that a high soil particle load of the feedstock materials can result in a lower percentage OM on dry weight. The pH range of the vegetable based composts includes lower pH values, while the pH range of the composts based on farmyard manure includes higher values. The C/N and C/P ratios of the vegetable based composts are considerably higher compared to the manure based composts from our experiments. Further, the C/P and N/P ratios of the solid fraction composts are the lowest. Consequently, with the vegetable based composts from this study, more C per unit of N and P can be added compared to manure based composts, while with composts from solid fraction less C and N per unit of P can be applied compared to vegetable and farmyard manure based composts. As such, the vegetable based composts from this study are more suited for increasing the SOM content, while the manure containing composts (with a higher total nutrient content (N, P, K) and mineral N content and with a lower C/N and C/P ratio) are preferred for use as slow-release fertilizer. However, adding proportionally more C-rich bulking agents to the manure, C/N and C/P ratios of manure based composts would increase.

However, a high C/N ratio (> 20-25) and consequently high C/P ratio or too high NH_4^+-N concentrations (> 400 mg kg^{-1} DM), can indicate a still biodegradable, not stabilized compost which is undesirable for crop growth (Bernal et al., 1998). Consequently, also stability parameters such as OUR and biodegradation potential and/or the pile temperature at the end of the composting process should be taken into account to assess the compost quality and purpose. Figure 8.1 shows that 75% of the biodegradation potential values fall below 2.8. Composts with a biodegradation potential < 1.8 are considered as stable (Blanco and Almendros, 1997; Eklind and Kirchmann, 2000; Francou et al., 2008; Lashermes et al., 2012;

Vandecasteele et al., 2014). Similarly, 75% of the OURs fall below 15 mmol kg^{-1} OM h^{-1}, the Flemish norm for stabilized composts.

8.1.4.2. Biodegradation potential

Throughout the different experiments, we used the biodegradation potential, i.e., the ratio of the (hemi-)cellulose content on the lignin content, as a parameter for estimating the biochemical stability of the feedstock materials and composts. The higher the value, the higher the biochemical degradability of the material. The parameter is not commonly used in Flanders, however, several published research already showed the potential of this parameter as a suitable stability parameter. We have performed a linear regression analysis between the biodegradation potential and the OUR (R, version 3.1.3; R CoreTeam, 2015), a commonly accepted stability parameter for compost used in Flanders, for all compost data (both feedstock and end product) obtained during this PhD research (n = 172). For the composted solid fraction, we concluded that OUR was not a suitable stability parameter, as very low OUR values were observed (Chapter 6). Therefore, the data of the solid fraction composts were not included in the regression analysis. When plotting the biodegradation potential versus the OUR for the feedstock materials before composting (n = 95), we found no significant relationship (data not shown). When testing this for the stable end products after composting (n = 61), we found a positive linear relationship between biodegradation potential and OUR (R^2 = 0.35, p < 0.01, slope (± s.e.) = 0.09 (± 0.02)) (Figure 8.2). Adding a factor accounting for the compost type significantly improved the regression model in terms of R^2 (R^2 = 0.69, p < 0.01, slope (± s.e.) = 0.15 (± 0.02)). There was an effect of compost type: for an equal biodegradation potential, cattle farmyard manure based composts had a lower OUR (estimated coefficient = -1.14, s.e. = 0.17, p < 0.01) than vegetable based composts. Adding an interaction term between OUR and compost type could not significantly improve the model (p = 0.40), indicating that there was no difference in the slope of the regression between both compost types.

In general, we can conclude that the lower the OUR after composting, i.e., the more stabilized the compost, the lower the biodegradation potential. Additionally, we have proved that the biodegradation potential decreased with time (Chapter 4), consequently, we carefully conclude that this parameter can be used to monitor the stabilization process during and after composting.

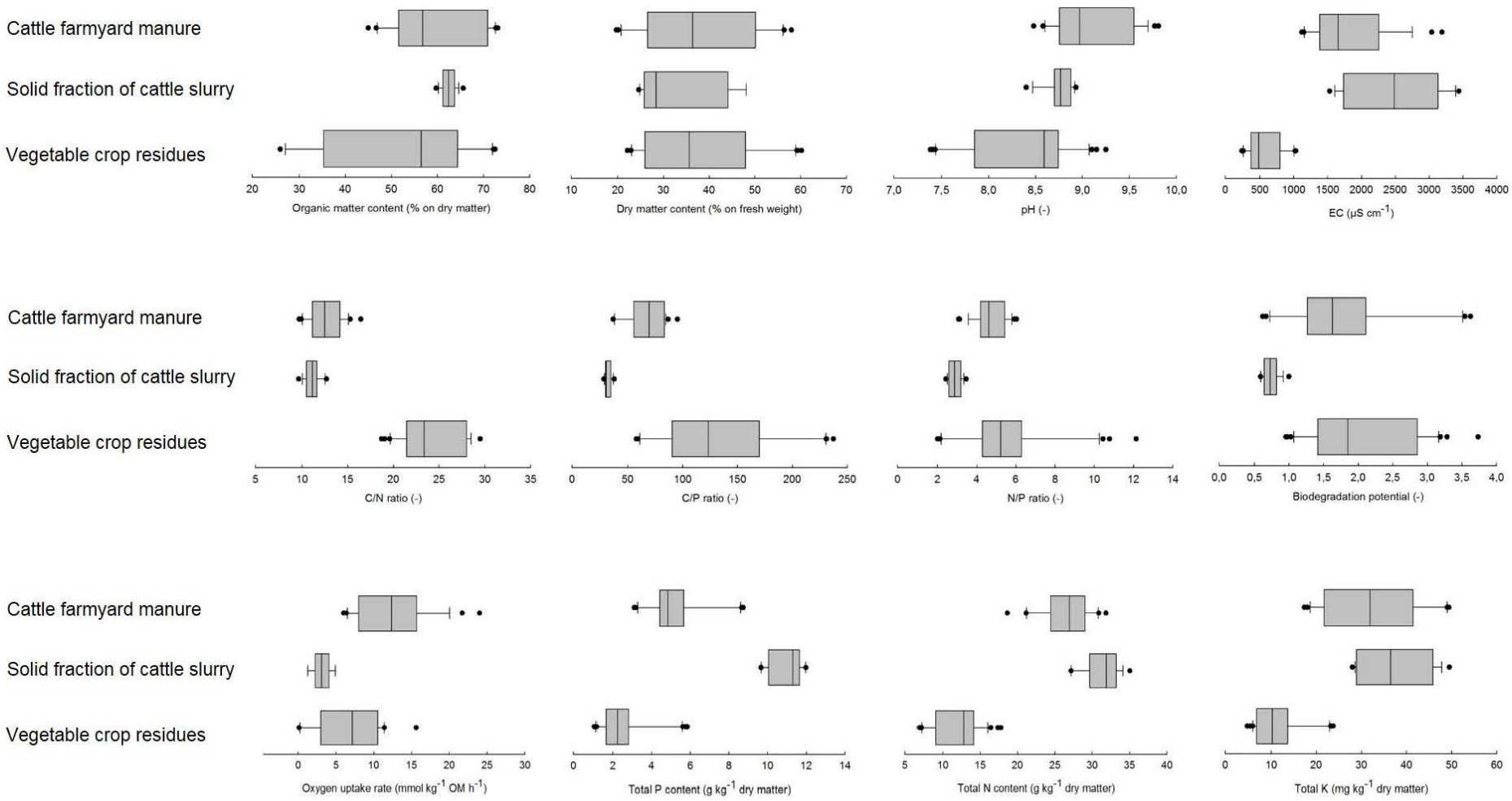


Figure 8.1 Compost parameters from the end products of the compost experiments with cattle farmyard manure (*n* = 29), solid fraction of cattle slurry (*n* = 16) and vegetable crop residues (*n* = 32), conducted during this PhD research. Box plots show minimum values (left whisker), first quartiles (left vertical line), median values (middle vertical line), third quartiles (right vertical line), maximum values (right whisker) and outlier values (black dots).

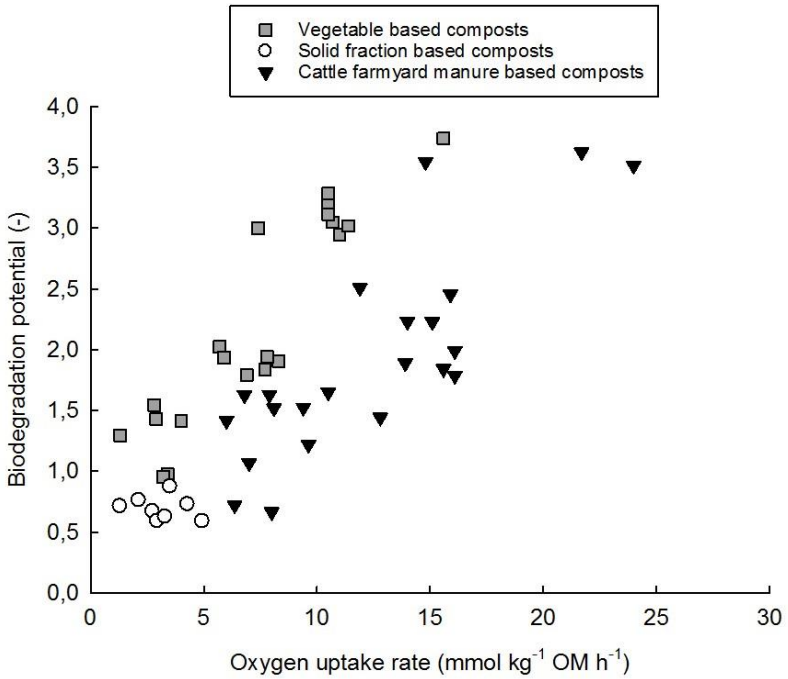


Figure 8.2 Biodegradation potential versus oxygen uptake rate for the stable composts.

8.1.4.3. Near infrared spectroscopy

Characterization of chemical properties and stability of feedstock materials and end products of composting allows for better process control and focused compost application, but lab-based analyses are time-consuming and thus expensive. Near infrared spectroscopy (NIRS) has been reported to allow for fast screening of chemical and biochemical properties of compost (Galvez-Sola et al., 2010a; Galvez-Sola et al., 2010b) and soil (Terhoeven-Urselmans et al., 2008). Two datasets were used for testing the use of NIRS for assessing chemical properties and stability of feedstock materials and composts, i.e., dataset 1 with 149 samples from feedstock materials, feedstock mixtures and composts and silages based on vegetable crop residues (including the data from Chapters 3 and 4 and from Agneessens et al. (2015)) and dataset 2 with 230 samples from feedstock materials and composts of 48 full-scale compost trials at operational compost facilities with green waste (Vandecasteele et al., 2016). Both datasets were chemically characterized as described in Chapter 4. Dry and ground samples were scanned with a FOSS XDS monochromator instrument with ISIscan v2.85.1 software. The inverse reflectance (log (1/R)) was measured from 400 to 2500 nm in steps of 2 nm. The samples were scanned in duplicate and the spectra were averaged. On the

basis of database 1, calibration development was executed with WINSI v4.9.0. software using modified partial least squares regression (Shenk and Westerhaus, 1991). Cross-validation was used to select the optimum number of partial least squares (PLS) terms. Depending on the measured characteristic, one or more samples identified as outlier were removed by the software and not included in the calibration. The standard error of calibration (SEC), standard error of cross-validation (SECV), the determination coefficient (R^2) of the simple linear regression between reference values and NIRS predicted values of the calibration set were calculated. Finally the ratio of the standard deviation of the calibration data set to the standard error of cross-validation (SD/SECV), indicating the performance of the calibrations, was calculated. On database 2, the standard error of prediction corrected for bias (SEP(C)) was calculated on the basis of the calibration equations derived from database 1.

The calibration results for dataset 1 with NIRS indicate that the potential to use NIRS to assess the chemical properties and stability of feedstock materials and composts based on vegetable crop residues is high, as high R^2 and acceptable SECV values were observed. The SD/SEVC ratio ranged from 2.6 for NDF to 5.9 for pH-H_2O and biodegradation potential which indicates the potential to use NIRS. However, validation of this relation with dataset 2 indicated that the observed correlation for dataset 1 was not valid for dataset 2, as the SEP (C) exceeded the limit (SEC*1.3) for all parameters, and the slopes had values deviating strongly from the intended value of 1. When the two different datasets were merged, the calibration results indicate a high potential for using NIRS as again high R^2 (> 0.90), acceptable SECV values (0.30-7.69) and acceptable SD/SEVC ratios (2.6-5.0) were observed (Table 8.2). This illustrates that for both datasets with specific characteristics each a successful calibration curve was fitted by the software. Altogether these results indicate that NIRS might be applicable for fast screening of chemical properties (OM, pH, C/N ratio, P and K content) and compost stability (OUR and biodegradation potential) during the process. Using NIRS resulted in time- and cost-efficient assessment of compost characteristics, allowing the operator for a faster quality assessment at the set-up of the windrows, and at the final stage of the process. Determining the stability by the OUR and cell wall components is time and cost-intensive (those measurements lasts for more than 1 week and need more than 8h of labor), consequently, the stability could be assessed much faster (in a few minutes) by NIRS. However, calibration should be updated regularly based

on new chemical analyses to ensure the validity of the calibration curve for other types of compost.

Table 8.2 Calibration results for the merged dataset 1 and 2. With *n* = the amount of samples used for calibration, SEC = standard error of calibration, SECV = standard error of cross-validation, SD = standard deviation of the calibration.

Parameter	*n*	Average	Min.	Max.	SEC	R^2	SECV	SD	SD/SECV
OM (% of DM)	356	61.2	23.1	95.6	3.65	0.97	4.22	21.30	5.0
pH-H_2O	225	7.7	3.7	9.6	0.24	0.97	0.30	1.45	4.8
NDF (% of DM)	280	43.6	15.0	77.4	3.72	0.95	4.37	16.10	3.7
ADF (% of DM)	281	32.8	12.6	60.7	2.81	0.95	3.56	12.10	3.4
ADL (% of DM)	282	14.7	3.2	27.5	1.49	0.92	1.69	5.38	3.2
Hemicellulose (% of DM)	279	9.3	1.3	26.1	1.49	0.92	1.99	5.51	2.8
Cellulose (% of DM)	279	15.7	1.3	41.3	2.69	0.93	3.39	9.96	2.9
Lignin (% of DM)	282	13.1	1.3	27.5	2.11	0.90	2.55	6.75	2.6
Biodegradation potential (-)	274	2.0	0.8	11.4	0.41	0.90	0.46	1.83	4.0
C/N (-)	354	29.1	9.4	139.0	3.16	0.96	3.65	17.60	4.8
OUR (mmol O_2 kg^{-1} OM h^{-1})	207	18.7	0.9	143.0	6.28	0.94	7.69	27.60	3.6
Total P (g kg^{-1} DM)	291	2.3	0.3	8.0	334.0	0.96	0.40	1.63	4.1
Total K (g kg^{-1} DM)	291	12.6	1.7	47.7	1829.0	0.96	2.25	9.46	4.2

8.2. Socio-economic factors hindering on-farm composting and the opportunities of alternative business scenarios

Three different action research cases, in which on-farm composting was performed in an alternative, collaborative form, led to new insights in how alternative business scenarios may be useful to overcome barriers concerning product quality, costs and legislation (Chapter 7).

Currently, buying compost from an industrial composting facility is still cheaper than on-farm production of compost. However, the environmental benefits from e.g. recycling of farm residues are not taken into account in this comparison as their pay-off is often on the long-term and the monetary value of these benefits is difficult to estimate (Obersteiner and Linzner, 2007). Nevertheless, Chapter 7 has shown that cooperation among farmers seems to lower production costs and can be a first step to increase the feasibility of on-farm composting. When paying a service provider to perform the composting process, a high amount of biomass should be available for processing all at once (as in case study 1 from Chapter 7) to make it economically advantageous. In general, hiring the required composting equipment is always cheaper than buying it, also in the long term. However, this may be very farm-specific and is likely to be scale-dependent.

In the three action cases, we could clearly observe that the complex and not-stimulating regulation is a major barrier for on-farm composting. Especially, the restrictions in the use of off-farm, external biomass and the related environmental license, including the requirement for a concrete pad to compost on, and the inspections by certification bodies are discouraging farmers to compost. Especially since the results of Chapter 7 clearly showed that the cooperation between farmers and third parties for exchanging biomass resulted in an end product of higher quality, than when applying or valorizing only the own byproducts. Furthermore, adding bulking agents from nature areas resulted in less N leaching than when only composting farmyard manure (Chapter 5). We assume that extending the definition of on-farm composting, which is now restricted to one farmer composting his or her own organic-biological residues and applying the compost product on his or her own fields, would stimulate on-farm composting. More specifically, one option would be to exclude farmers, composting relatively small amounts on occasional basis and for own use, from the environmental license requirement when using external biomass from nature areas. In the Netherlands for example, grass clippings from nature areas can be used in on-farm composting without extra regulations when the clippings have a heavy metal and micropollutant contents below the standards, and are transported over a limited distance[11]. Further, we think that the cooperation within a small and clearly delineated group of nearby farms to compost their byproducts could be considered as on-farm composting and the compost could be considered as a natural product from the farm. In that way, locally available byproducts, too small in volume for other valorizations, are valorized on a local scale and the compost can be exchanged between farms without extra financial investments and administrative burdens. Further, there would be no competition with industrial composting facilities because the trade of compost is limited between the farmers. Of course, restrictions on the type of biomass, amount of compost production, transport distance, etc. should be defined.

Besides cooperation, other strategies can be used to create incentives for on-farm composting. For example, research and demonstration projects, extension educational programs, financial grants and low-interest loans have succeeded in expanding the practice of composting among farms in the US (Kashmanian and Rynck, 1998).

[11] http://www.rwsleefomgeving.nl/onderwerpen/afval/groenafval/maaisel/

8.3. Lessons learned from the innovative research approach

Optimal valorization of byproducts and sustainable use and management of natural resources are key points in the transition to a sustainable bio-economy and agricultural systems, in which agro-ecological practices are important. Local on-farm composting of byproducts and farm compost application can play an important role in this, even if rarely recognized. On the one hand, biomass residues are valorized, while on the other hand, compost application contributes to a healthy soil, the basic resource for biomass production. However, despite extensive research on general composting techniques, and well established positive effects of compost application on soil quality, the adaptation of on-farm composting in practice is minimal in Flanders. The results of the study described in Chapter 2 reveal that other factors (e.g., of institutional, market, financial and/or behavioral nature), play a role in hindering on-farm composting and compost application. In other words, research on the scientific and technological aspects of on-farm composting only, would not be sufficient to stimulate on-farm composting in practice. Therefore, in order to actually (i) gain insight in the (hindering and stimulating) factors at play in practice, and (ii) achieve a breakthrough of innovative ways of on-farm composting and compost use, another approach of the research is currently needed. Throughout the GENESYS project (described in Section 1.1), an open innovation approach and a set of guiding principles for the management of this process were followed, as described by Van Lancker et al. (2016). In the commonly developed GENESYS framework, cooperation with relevant stakeholders is a key issue to improve the implementation of the innovation in practice. In this thesis, we did this through stakeholder consultations and involvement in action research cases. Defoer et al. (1998) already showed that combining quantitative analysis and participatory action research can lead to joint learning between researchers and farmers, leading to planning, experimenting and adapting ways to improve the use of scarce local resources.

In this thesis, the transdisciplinary (interdisciplinary and participatory) approach enabled us to **further develop a broad network of relevant stakeholders**, building on the already existing network from previous research and extension activities at ILVO. This stakeholder network is spanning the entire compost value chain, including:

- The supply side: farmers, nature conservation and landscape managers who produce byproducts

- The processors: industrial composting facilities, service providers, contractors
- The demand side: farmers who want to use compost
- All affected by policy and regulations

The network allowed us to:

(1) **Identify problems and challenges** for on-farm composting and compost application. The problems and challenges linked to on-farm composting already known by the ILVO research team and the scientific community were confirmed (e.g., the shortage in woody biomass to make an optimal feedstock mixture) and new challenges were identified (e.g., the high number of leased fields hindering farmers from investing in soil quality through compost application was highlighted). Further, it became more clear which hindering factors had more impact than others for the practitioners.

(2) **Develop research questions** that were **both scientifically valuable and practically feasible**. These research questions were tuned to scientifically relevant questions, while guaranteeing that the results are useful in practice. Because one or more challenges were tackled or the research approach took into account certain practical prerequisites, we could find feasible (from an economic or practical viewpoint) solutions to valorize agro-food byproducts.

(3) **Enable real life studies** besides lab and field scale experiments at research institutes. Through the broad stakeholder network we could initiate action research cases, in which supply chain members performed the innovation in practice, with support and supervision of a research institute. Furthermore, we were able to perform compost experiments with feedstock materials accessible from the stakeholders (e.g., spent growth media from tomato cultivation).

(4) **Provide insights in the non-scientific aspects** of on-farm composting and application. Real-life circumstances were approached in the action research cases, which helped us to gain more insight in both technical and practical challenges (e.g. economic feasibility under different scenarios) compared to lab scale studies. The (theoretically) identified hindering factors became visible in real-life, which helped us to even understand them better.

(5) **Create a larger legitimacy** for the research and advocates for on-farm composting. By directly communicating with stakeholders and actively involving them during the research, we could create a broader support and awareness for on-farm composting. In

addition, the mutual cooperation between the stakeholders in the action research cases resulted in a peer-to-peer network by which the stakeholders could e.g., convince each other from the value of on-farm composting. In this way, certain misconceptions from the practitioners (e.g. "application of compost originating from byproducts from nature management always results in weed problems") could be adjusted and framed by scientifically and experienced-based advice. Additionally, all three action research cases continued with on-farm composting after the project, indicating the positive impact of performing action research in participation with stakeholders on the acceptance of the innovation. By this, we created advocates for on-farm composting, who will further increase the awareness for on-farm composting.

(6) **Disseminate the research results** faster and easier. By the network and the close contact with practitioners and policy advisors, we had a direct outlet for the research results, leading to the preparation of policy notes. For example, a policy note was written in collaboration with several interest groups to the OVAM to ease on-farm composting. More specifically, it was suggested to extend the definition of on-farm composting, now restricted to one farmer who composts own biomass and applies compost on own fields: (1) the use of extern biomass from nearby nature reserves should be possible without the requirement for an environmental license, and (2) the cooperation between a restricted amount of nearby farms to compost their byproducts should be subjected to less strict environmental regulations. In that way, locally available byproducts, too small in volume for other valorizations, are valorized on a local scale and the compost can be exchanged between farms. Furthermore, partly based on the results of our research (described in Chapter 5), regulations for field storage of solid manure were adapted in the fifth Manure Action Plan: field storage was prolonged from one to two months between 15/01 and 15/11.

(7) **Increase the recognition and network of ILVO**. The stakeholder network makes it easier to make use of knowledge and equipment from the stakeholders, that is not available at ILVO. In addition, those actors can be partners in other research projects of ILVO and improves the chance that the actors contact ILVO as partner in their research projects.

8.4. Suggestions for future research

This thesis clearly shows opportunities to improve on-farm composting and compost application in Flemish agriculture. However, based on the stakeholder participation and technological research, some new research questions, both technological and socio-economic, arose and can be addressed in future research.

8.4.1. Technological research

A first suggestion for future research, is focusing on the evaluation of nutrient losses, both liquid and volatile, throughout the entire management chain: from stable or field, over storage, spreading on the field and further degradation in the soil. This would contribute to environmental decision making on best practices for management of byproducts. To estimate total nutrient losses during processing, mass balances are needed. However, from this and other studies, it can be concluded that there are large variations on the outcomes of mass balances of a field study, because errors occur during transport, weighing, turning the pile, taking samples, analyses, etc. Shah (2013) did an attempt to quantify N losses during the whole management chain of solid cattle manure and concluded that individually applied mitigation practices often result in compensatory loss pathways. For instance, methods that reduced ammonia emissions during storage (e.g., tight covering the manure pile with a plastic as we did in Chapters 5 and 6) resulted in larger losses after surface application of manure to the field. In addition, a full environmental assessment of different management options on the long-term should be taken into account in such an analysis. However, it is difficult to translate environmental benefits into assessment methodologies such as Life Cycle Assessment (LCA) (Obersteiner & Linzner, 2007). Cycle closure, C sequestration and the related positive effects on the emission of greenhouse gases, water holding capacity and improvement of soil structure are not taken into account in the majority of the cases. This is due to the fact that, to this date, no widely accepted methods have been identified to cover those benefits by common impact categories of LCA.

Related to this, another aspect that could be further investigated is P leaching during different management options of cattle farmyard manure, as we only focused on N leaching (Chapter 5). We observed a decreased risk for N leaching during co-composting farmyard manure with bulking agents in comparison with composting manure only (Chapter 5), therefore, it could be assessed if P losses would decrease as well.

With regard to the application of composts, silages and digestates, the effect on soil C and nutrient dynamics is of major importance. Our findings (Chapters 3 and 6) were limited to incubation (N and C mineralization, N_2O emissions) and pot experiments and should be further validated with results from field trials under different conditions (e.g. soil type and crop production) and with repeated application over a longer time period. For example, according to Thomsen et al. (2013), the C retention in soil over decades to centuries appears to be similar whether the initial turnover of plant biomasses occurs in the soil, in the digestive tract of ruminants, in an anaerobic reactor or in a combination of the latter two. With regard to nutrient dynamics, it was recently shown that soil application of plant based farm compost could result in less P leaching than when applying raw manure (Vanden Nest et al., 2016). Further, D'Hose et al. (2016) demonstrated that application of plant based farm compost for 4 years, on top of cattle or pig slurry application, did not induce higher N and P leaching. Another important aspect for soil amendment is the hygienisation of the amended product. Composting leads to destruction of pathogens (Lung et al., 2001) and weed seeds (Eghball and Lesoing, 2000) and is therefore an excellent technique to sanitize biomass for export. However, it could be interesting to test the survival of pathogens and weed seeds during ensiling of crop residues and manure. Studies on maize found that ensiling can limit *Fusarium* species due the low oxygen content and acid pH, however other fungi can survive the silage environment (Mansfield and Kuldau, 2007).

8.4.2. Socio-economic research

The estimation of the costs for on-farm composting in this research was only a first step in an economical comparison of the different valorization options for biomass. Extra research is required to obtain more accurate parameters to calculate the revenues and costs of on-farm composting. Furthermore, a more extensive economic study could provide more insights in the effect of scale for on-farm composting, but also compare different business scenarios and/or examine the impact of more stimulating legislation on on-farm composting. Such studies require a more holistic approach, taking into account detailed economic (logistic costs, supply and demand of byproducts, interaction with other sectors, etc,), social (e.g. relationships among farmers) and environmental (e.g. weather) factors which were out of the scope of this research.

The timescale of this study was limited to the short-term benefits of on-farm composting. Obviously, systematic composting of farm residues and compost application may also result in long-term environmental benefits and ecosystem services which are important in policy

decisions. However, more research is needed to estimate, and valuate these (potential) benefits and services, as well as the ecological-economic trade-offs.

References

Aggelides, S. M. and P. A. Londra, 2000. Effects of compost produced from town wastes and sewage sludge on the physical properties of a loamy and a clay soil. *Bioresource Technology*, 71, 253-259.

Agneessens, L., Viaene, J., Vanden Nest, T., Vandecasteele, B. and De Neve, S., 2015. Effect of ensilaged vegetable crop residue amendments on soil carbon and nitrogen dynamics. *Scientia Horticulturae*, 192, 311-319.

Agneessens, L., De Waele, J. and De Neve, S., 2014. Review of alternative management options of vegetable crop residues to reduce nitrate leaching in intensive vegetable rotations. *Agronomy*, 4, 529-555.

Aguerre, M. J., Wattiaux, M. A., Hunt, T. and Lobos, N. E., 2012. Effect of nitrogen content and additional straw on changes in chemical composition, volatile losses, and ammonia emissions from dairy manure during long-term storage. *Journal of Dairy Science*, 95, 3454-3466.

Ajzen, I., 1991. The Theory of Planned Behavior. *Organizational Behavior and Human Decision Processes*, 50, 179-211.

Ajzen, I., 1988. Attitudes, personality and behavior. Dorsey Press, Chicago: IL.

Alburquerque, J. A., Gonzálvez, J., García, D. and Cegarra, J., 2006. Effects of bulking agent on the composting of "alperujo", the solid by-product of the two-phase centrifugation method for olive oil extraction. Process Biochem. 41, 127-132.

Ameloot, N., Maenhout, P., De Neve, S. and Sleutel, S., 2016. Biochar-induced N2O emission reductions after field incorporation in a loam soil. *Geoderma*, 267, 10-16.

Amlinger, F., Gotz, B., Dreher, P., Geszti, J. and Weissteiner, C., 2003. Nitrogen in biowaste and yard waste compost: dynamics of mobilisation and availability - a review. *European Journal of Soil Biology*, 39, 107-116.

Annabi, M., Houot, S., Francou, F., Poitrenaud, M., and Le Bissonnais, Y., 2007. Soil aggregate stability improvement with urban composts of different maturities. *Soil Science Society of America Journal*, 71, 413-423.

Baggs, E. M., Rees, R. M., Smith, K. A. and Vinten, A. J. A., 2000. Nitrous oxide emission from soils after incorporating crop residues. *Soil Use Manage*, 16, 82-87.

Barrington, S., Choinière D., Trigui, M. and Knight, W., 2002. Effect of carbon source on compost nitrogen and carbon losses. Bioresource Technology, 83, 189-194.

Barth, J., Amlinger, F., Favoino, E., Siebert, S., Kehres, B., Gottschal, R., Bieker, M., Löbig, A. and Bidlingmaier, W., 2008. Compost production and use in the EU. Final report. European Compost Network. 182p.

Bernal, M. P., Alburquerque, J.A. and Moral, R., 2009. Composting of animal manures and chemical criteria for compost maturity assessment. A review. *Bioresource Technology*, 100, 5444-5453.

Bernal, M. P., Paredes, C., Sanchez-Monedero, M. A. and Cegarra, J., 1998. Maturity and stability parameters of composts prepared with a wide range of organic wastes. *Bioresource Technology*, 63, 91-99.

Bijttebier, J. et al., 2015. Farmers review of Best Management Practices: drivers and barriers as seen by adopters and non-adopters. Compatibility of Agricultural Management Practices and Types of Farming in the EU to enhance Climate Change Mitigation and Soil Health (CATCH-C). EU FP7 N° 289782. 171p.

BioForum Vlaanderen vzw, 2013. Compostbrochure. Agro-ecologische aanpak op jouw bedrijf. 25p.

Blanco, M. J. and Almendros, G., 1997. Chemical transformation, phytotoxicity and nutrient availability in progressive composting stages of wheat straw. *Plant and Soil*, 196, 15-25.

Bokhorst, J. and ter Berg, C., 2001. Handboek mest & compost: behandelen, beoordelen en toepassen, Louis Bolk Instituut.

Breitenbeck, G. A. and Schellinger, D., 2004. Calculating the Reduction in Material Mass And Volume during Composting. *Compost Science & Utilization*, 12, 365-371.

Bremner, J. M., 1997. Sources of nitrous oxide in soils. *Nutrient Cycling in Agroecosystems*, 49, 7-16.

Brito, L., Mourão, I., Coutinho, J. and Smith, S., 2012. Simple technologies for on-farm composting of cattle slurry solid fraction. *Waste Management*, 32, 1332-1340.

Brito, L., Coutinho, J. and Smith, S., 2008. Methods to improve the composting process of the solid fraction of dairy cattle slurry. *Bioresource Technology*, 99, 8955-8960.

Bustamante, M., Alburquerque, J.A., Restrepo, A., de la Fuente, C., Paredes, C., Moral, R. and Bernal, M., 2012. Co-composting of the solid fraction of anaerobic digestates, to obtain added-value materials for use in agriculture. *Biomass & Bioenergy*, 43, 26-35.

Caceres, R., Flotats, X. and Marfa, O., 2006. Changes in the chemical and physicochemical properties of the solid fraction of cattle slurry during composting using different aeration strategies. *Waste Management*, 26, 1081-1091.

Cayuela, M. L., Sanchez-Monedero, M. A. and Roig, A., 2010. Two-phase olive mill waste composting: enhancement of the composting rate and compost quality by grape stalks addition. *Biodegradation*, 21, 465-473.

Chadwick, D.R., 2005. Emissions of ammonia, nitrous oxide and methane from cattle manure heaps: effect of compaction and covering. *Atmospheric Environment*, 39, 787-799.

Chang, J. I. and Chen, Y., 2010. Effects of bulking agents on food waste composting. *Bioresource Technology*, 101, 5917-5924.

Chaves, B., De Neve, S., Piulats, L. M., Boeckx, P., Van Cleemput, O. and Hofman, G., 2007. Manipulating the N release from N-rich crop residues by using organic wastes on soils with different textures. *Soil Use Management*, 23, 212-219.

Cleary, J., Roulet, N. T. and Moore, T. R., 2005. Greenhouse Gas Emissions from Canadian Peat Extraction, 1990-2000: A Life-cycle Analysis. *AMBIO: A Journal of the Human Environment*, 34, 456-461.

Congreves, A. K., Vyn, J. R. and Van Eerd, L. L., 2013. Evaluation of Post-Harvest Organic Carbon Amendments as a Strategy to Minimize Nitrogen Losses in Cole Crop Production. *Agronomy*, 3, 181-199.

Creswell, J. W., 2003. Research design: Qualitative, quantitative, and mixed method approaches, 2nd edition. SAGE Publications.

CUMA, 2015. Le réseau des Coopératives d'Utilisation de Matériel Agricole. Available online: http://www.cuma.fr/ (accessed on 27.01.2015).

Curtis, M. J. and Claassen, V.P., 2009. Regenerating Topsoil Functionality in Four Drastically Disturbed Soil Types by Compost Incorporation. *Restoration Ecology*, 17, 24-32.

Danso, G., Drechsel, P., Fialor, S. and Giordano, M., 2006. Estimating the demand for municipal waste compost via farmers' willingness-to-pay in Ghana. *Waste Management*, 26, 1400-1409.

Defoer, T., De Groote, H., Hilhorst, T., Kanté, S. and Budelman, A., 1998. Participatory action research and quantitative analysis for nutrient management in southern Mali: a fruitful marriage? Agriculture, Ecosystems and Environment, 71, 215-228.

de la Fuente, C., Alburquerque, J.A., Clemente, R. and Bernal, M., 2013. Soil C and N mineralisation and agricultural value of the products of an anaerobic digestion system. *Biology and Fertility of Soils*, 49, 313-322.

Delief, A. and De Vocht, A., 2012. Eindrapport Graskracht. Werkpakket 2: Inventarisatie. PHL BioResearch, 46p.

De Meester, S., C. Callewaert, E. De Mol, H. Van Langenhove, and Dewulf, J., 2011. The resource footprint of biobased products: a key issue in the sustainable development of biorefineries. *Biofuels Bioproducts & Biorefining*, 5, 570-580.

De Neve, S., Saez, S. G., Daguilar, B. C., Sleutel, S. and Hofman, G., 2004. Manipulating N mineralization from high N crop residues using on- and off-farm organic materials. *Soil Biology & Biochemistry*, 36, 127-134.

De Neve, S., Sleutel, S. and Hofman, G., 2003. Carbon mineralization from composts and food industry wastes added to soil. *Nutrient Cycling in Agroecosystems*, 67, 13-20.

De Neve, S. and Hofman, G., 1996. Modelling N mineralization of vegetable crop residues during laboratory incubations. *Soil Biology & Biochemistry*, 28, 1451-1457.

de Ruijter, F.J., ten Berge, H.F.M. and Smit, A.L., 2010. The Fate of Nitrogen from Crop Residues of Broccoli, Leek and Sugar Beet. *Acta Horticulturae*, 852, 157–162.

De Tender, C. A., Debode, J., Vandecasteele, B., D'Hose, T., Cremelie, P., Haegeman, A., Ruttink, T., Dawyndt, P. and Maes, M., 2016. Biological, physicochemical and plant health responses in lettuce and strawberry in soil or peat amended with biochar. *Applied Soil Ecology*, 107, 1-12.

Denzin, N., 2006. Sociological methods: A sourcebook. 5th edition, Aldine Transaction.

Dewes, T., Ahrens, E. and Koch, C., 1993. Nitrogen penetration and persistence in soil under uncovered and covered farmyard manure heaps. *Journal of Agronomy and Crop Science-Zeitschrift fur Acker und Pflanzenbau*? 170, 73-83.

D'Hose, T., Ruysschaert, G., Viaene, N., Debode, J., Vanden Nest, T., Van Vaerenbergh, J., Cornelis, W., Willekens, K. and Vandecasteele, B., 2016. Farm compost amendment and non-inversion tillage improve soil quality without increasing the risk for N and P leaching. *Agriculture, Ecosystems & Environment*, 225, 126-139.

D'Hose, T., Cougnon, A., De Vliegher, A., Vandecasteele, B., Viaene, N., Cornelis, W., Van Bockstaele, E., and Reheul, D., 2014. The positive relationship between soil quality and crop production: A case study on the effect of farm compost application. *Applied Soil Ecology*, 75, 189-198.

Diacono, M. and Montemurro, F., 2010. Long-term effects of organic amendments on soil fertility. A review. *Agronomy for Sustainable Development*, 30, 401-422.

Diara, C., Incrocci, L., Pardossi, A. and Minuto, A., 2012. Reusing Greenhouse Growing Media. *Acta Horticulturae*, 927, 793-800.

Dias, B. O., Silva, C. A., Higashikawa, F. S., Roig, A. and Sanchez-Monedero, M. A., 2010. Use of biochar as bulking agent for the composting of poultry manure: Effect on organic matter degradation and humification. *Bioresource Technology*, 101, 1239-1246.

Dondeyne, S., Vanierschot, L., Langohr, R., Van Ranst, E. and Deckers, S., 2014. The soil map of the Flemish region converted to the 3rd edition of the World Reference Base for soil resources (41 map sheets at scale 1:40 000, 1 map sheet at 1:250 000). KU Leuven, Universiteit Gent, Vlaamse Overheid, Brussels.

Driehuis, F., Lucas-van den Bos, E., Bomhof, M. H. J. and Wells-Bennik, M. H. J., 2015. Sporen van thermofiele sporenvormers in gescheiden mest, gecomposteerde houtsnippers en andere beddingmaterialen. (Spores of thermophilic aerobic sporeformers in manure solids, composted wood chips and other bedding materials). NIZO-Rapport E 2015/040. NIZO Food Research BV, Ede, The Netherlands. 34p.

Driehuis, F., Lucas-van den Bos, E., Wagendorp, A., and Wells-Bennik, M. H. J., 2014. Sporen van thermofiele aërobe sporenvormers in compost en andere beddingmaterialen bij melkveebedrijven met een vrijloop- of ligboxenstal. (Spores of thermophilic aerobic sporeformers in compost and other bedding materials used by dairy farmers with a bedded pack or freestall barn). NIZO-Report E 2014/045. NIZO Food Research BV, Ede, The Netherlands. 37p.

Eghball, B. and Lesoing, G. W., 2000. Viability of weed seeds following manure windrow composting. *Compost Science & Utilization*, 8, 46-53.

Eghball, B., Power, J. F., Gilley, J. E. and Doran, J. W., 1997. Nutrient, carbon, and mass loss during composting of beef cattle feedlot manure. *Journal of Environmental Quality*, 26, 189-193.

Eklind, Y. and Kirchmann, H., 2000. Composting and storage of organic household waste with different litter amendments. I: carbon turnover. *Bioresource Technology*, 74, 115-124.

El Kader, N. A., Robin, P., Paillat, J.M. and Leterme, P., 2007. Turning, compacting and the addition of water as factors affecting gaseous emissions in farm manure composting. *Bioresource Technology*, 98, 2619-2628.

European Commission, 2012. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committees of the Regions. Innovating for Sustainable Growth: A Bioeconomy for Europe, (SWD(2012) 11 final).

European Commission, 2006. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee of the Regions. Thematic Strategy for Soil Protection, (COM(2006) 231 final).

Faber, A. and Hoppe, T., 2013. Co-constructing a sustainable built environment in the Netherlands-Dynamics and opportunities in an environmental sectoral innovation system. *Energy Policy*, 52, 628-638.

Fabrizio, A., Tambone, F. and Genevini, P., 2009. Effect of compost application rate on carbon degradation and retention in soils. *Waste Management*, 29, 174-179.

Ford, M. and Fleming, R., 2002. Mechanical solid-liquid separation of livestock manure. Literature review. In: Report to Ontario Pork, Case Study 7 - Screw press. Ridgetown College, University of Guelph. 52p.

Foston, M. and Ragauskas, A. J., 2012. Biomass Characterization: Recent Progress in Understanding Biomass Recalcitrance. *Industrial Biotechnology*, 8, 191-208.

Francou, C., Linères, M., Derenne, S., Villio-Poitrenaud, M. L. and Houot, S., 2008. Influence of green waste, biowaste and paper-cardboard initial ratios on organic matter transformations during composting. *Bioresource Technology*, 99, 8926-8934.

Francou, C., Poitrenaud, M. and Houot, S., 2005. Stabilization of organic matter during composting: Influence of process and feedstocks. *Compost Science & Utilization*, 13, 72-83.

Fuchs, J. G., Baier, U., Berner, A., Mayer, J. and Schleiss, K., 2008. Effects of digestate on the environment and on plant production - results of a research project. Workshop at: ECN/ORBIT e.V. Workshop 2008 "The future for Anaerobic Digestion of Organic Waste in Europe", Germany, Nuremberg. 11p.

Galvez, A., Sinicco, T., Cayuela, M. L., Mingorance, M. D., Fornasier, F. and Mondini, C., 2012. Short term effects of bioenergy by-products on soil C and N dynamics, nutrient availability and biochemical properties. Agriculture, *Ecosystems & Environment*, 160, 3-14.

Garcia-Gomez, A., Bernal, M. P. and Roig, A., 2003. Carbon mineralisation and plant growth in soil amended with compost samples at different degrees of maturity. *Waste Management & Research*, 21, 161-171.

Godden, B. and Penninckx, M., 1997. Management of farmyard manure composting is important to maintain sustainability in organic farming. In: Isart J. and Llerena J.J. (eds), Resource use in organic farming. Proceedings of the third ENOF workshop, June 5-6, 1997, Ancona, Italy.

Gomez-Brandon, M., Lazcano, C. and Dominguez, J., 2008. The evaluation of stability and maturity during the composting of cattle manure. *Chemosphere*, 70, 436-444.

Grey, M., and Henry, C., 1999. Nutrient retention and release characteristics from municipal solid waste compost. *Compost Science & Utilization*, 7, 42-50.

Grigatti, M., Cavani, L. and Ciavatta, C., 2011. The evaluation of stability during the composting of different starting materials: Comparison of chemical and biological parameters. *Chemosphere*, 83, 41-48.

Gustavsson, J., Cederberg, C., Sonesson, U., van Otterdijk, R., and Meybeck, A., 2011. Global food losses and food waste - Extent, causes and prevention. Food And Agriculture Organization of the United Nations, Rome. 30p.

Gybels, R., Viaene, J., Vandervelden, J., Reubens, B. and Vandecasteele, B., 2013. Biomassa als bodemverbeteraar - Onderzoek naar de toepassing van beheerresten als bodemverbeteraar. Agentschap voor Natuur en Bos, Inverde & ILVO, 73p.

Hao, X. Y., Chang, C., Larney, F.J. and Travis, G.R., 2001. Greenhouse gas emissions during cattle feedlot manure composting. *Journal of Environmental Quality*, 30, 376-386.

Hedstrom, A., 2001. Ion exchange of ammonium in zeolites: A literature review. *Journal of Environmental Engineering*, 127, 673-681.

Huang, G. F., Wong, J. W. C., Wu, Q. T. and Nagar, B. B., 2004a. Effect of C/N on composting of pig manure with sawdust. *Waste Management*, 24, 805-813.

Huang, Y., Zou, J., Zheng, X., Wang, Y., Xu, X., 2004b. Nitrous oxide emissions as influenced by amendment of plant residues with different C:N ratios. *Soil Biology & Biochemistry*, 36, 973–981.

Hutchinson, M. and Griffin, T., 2008. Evaluation of fiber content relative to other measures of compost stability. *Compost Science & Utilization*, 16, 6-11.

Hutchison, M. L., Walters, L. D., Avery, S. M., Munro, F. and Moore, A., 2005. Analyses of livestock production, waste storage, and pathogen levels and prevalences in farm manures. *Applied and Environmental Microbiology*, 71, 1231-1236.

Inglezakis, V.J., Loizidou, M.M. and Grigoropoulou, H.P., 2004. Ion exchange studies on natural and modified zeolites and the concept of exchange site accessibility. *Journal of Colloid and Interface Science*, 275, 570-576.

Johnson, R. B., Onwuegbuzie, A.J. and Turner, L.A., 2007. Toward a Definition of Mixed Methods Research. *Journal of Mixed Methods Research*, 1, 112-133.

Kashmanian, R.M. and Rynk, R.F., 1998. Creating positive incentives for farm composting. American journal of alternative agriculture, 3, 40-45.

Keegan, D., Kretschmer, B., Elbersen, B. and Panoutsou, C., 2013. Cascading use: a systematic approach to biomass beyond the energy sector. *Biofuels Bioproducts & Biorefining*, 7, 193-206.

Keurcompost, 2014. Beoordelingsrichtlijn Keurcompost. Versie 3.0 - geldend vanaf 1 februari 2014.

Kirschenmann, F., 2010. Alternative agriculture in an energy- and resource-depleting future. *Renewable Agriculture and Food Systems*, 25, 85-89.

Kolenbrander, G.J. and De La Lande Cremer, L.C.N., 1967. Stalmest en gier: waarde en mogelijkheden. H. Veenman & Zonen N.V., Wageningen, The Netherlands.

Kristensen, H. L. and McCarty, G. W., 1999. Mineralization and immobilization of nitrogen in heath soil under intact Calluna, after heather beetle infestation and nitrogen fertilization. *Applied Soil Ecology*, 13, 187-198.

Krogmann, U. and Körner, I., 2000. Technology and strategies of composting. In: 'Environmental Processes III: Solid waste and waste gas treatment, preparation of drinking water'. (Ed. J Klein and J Winter) pp. 127-150. (Wiley-VCH: Weinheim).

Kuiters, A. T., 1990. Role of Phenolic Substances from Decomposing Forest Litter in Plant-Soil Interactions. *Acta Botanica Neerlandica*, 39, 329-348.

Lafrance, C., Lessard, P. and Buelna, G., 1996. Evaluation de la filtration sur tourbe et compost pour le traitement de l'effuent d'une usine de compostage de résidus verts. *Canadian Journal of Civil Engineering*, 23, 1041-1050.

Langlais, A., Nicourt, C., Bourblanc, M. and Gaigné, C., 2014. Livestock farming and nitrogen within the economic and social context. *Advances in Animal Biosciences*, 5, 20-27.

Larney, F. J., Olson, A. F., Miller, J. J., DeMaere, P. R., Zvomuya, F. and McAllister, T. A., 2008. Physical and chemical changes during composting of wood chip-bedded and straw-bedded beef cattle feedlot manure. *Journal of Environmental Quality*, 37, 725-735.

Larney, F. J. and Hao, X., 2007. A review of composting as a management alternative for beef cattle feedlot manure in southern Alberta, Canada. *Bioresource Technology*, 98, 3221-3227.

Lashermes, G., Barriuso, E., Le Villio-Poitrenaud, M. and Houot, S., 2012. Composting in small laboratory pilots: Performance and reproducibility. *Waste Management*, 32, 271-277.

Leconte, M. C., Mazzarino, M. J., Satti, P., Iglesias, M. C. and Laos, F., 2009. Co-composting rice hulls and/or sawdust with poultry manure in NE Argentina. *Waste Management*, 29, 2446-2453.

Lee, K. H., 2014. Drivers and Barriers to Energy Efficiency Management for Sustainable Development. *Sustainable Development*, 23, 16-25.

Lefcourt, A. M. and Meisinger, J. J., 2001. Effect of adding alum or zeolite to dairy slurry on ammonia volatilization and chemical composition. *Journal of Dairy Science*, 84, 1814-1821.

Li, Y. C., Stoffella, P. J. , Alva, A. K. , Calvert, D. V. and Graetz, D. A. , 1997. Leaching of nitrate, ammonium, and phosphate from compost amended soil columns. *Compost Science & Utilization*, 5, 63-67.

Lung, A. J., Lin, C. M., Kim, J. M., Marshall, M. R., Nordstedt, R., Thompson, N. P. and Wei, C. I., 2001. Destruction of Escherichia coli O157 : H7 and Salmonella Enteritidis in cow manure composting. *Journal of Food Protection*, 64, 1309-1314.

Maes, S., Elsen, A., Tits, M., Boon, W., Deckers, S., Bries, J., Vogels, N. and Vandendriessche, H., 2012. Wegwijs in de bodemvruchtbaarheid van de Belgische akkerbouw- en weilandpercelen (2008-2011). Bodemkundige Dienst van België. 198p.

Magri. A. and Teira-Esmatges, M.R., 2015. Assessment of a composting process for the treatment of beef cattle manure. *Journal of Environmental Science and Health Part B-Pesticides Food Contaminants and Agricultural Wastes*, 50, 430-438.

Mansfield, M. and Kuldau, G., 2007. Microbiological and molecular determination of mycobiota in fresh and ensiled maize silage. *Mycologia*, 99, 269-278.

Marcato, C. E., Mohtar, R., Revel, J. C., Pouech, P., Hafidi, M. and Guiresse, M., 2009. Impact of anaerobic digestion on organic matter quality in pig slurry. *International Biodeterioration & Biodegradation*, 63, 260-266.

Martins, O. and Dewes, T., 1992. Loss of Nitrogenous Compounds During Composting of Animal Wastes. *Bioresource Technology*, 42, 103-111.

Maschinenring, 2015. Available online: http://www.maschinenring.at/ (accessed on 27.01.2015).

Maxwell, D. and Zziwa, S., 1992. Urban farming in Africa: The case of Kampala. African Centre for Techology Studies, Nairobi, Kenya.

Meeusen, M. J.G., Hoogeveen, M.W. and Sengers, H.H.W.J.M, 1998. Groene reststromen in agroketens. Een beschrijving van de markt van organische reststromen uit de landbouw en de voedings- en genotmiddelenindustrie. Den Haag, Landbouw-Economisch Instituut (LEI-DLO), mededeling 608, 55p.

Meunchang, S., Panichsakpatana, S. and Weaver, R. W., 2005. Co-composting of filter cake and bagasse; by-products from a sugar mill. *Bioresource Technology*, 96, 437-442.

Meyer-Kohlstock, D., Haedrich, G., Bidlingmaier, W. and Kraft, E., 2013. The value of composting in Germany - Economy, ecology, and legislation. *Waste Management*, 33, 536-539.

Michel, F. C., Pecchia, J. A., Rigot, J. and Keener, H. M., 2004. Mass and nutrient losses during the composting of dairy manure amended with sawdust or straw. *Compost Science & Utilization*, 12, 323-334.

Miller, F.C., 1992. Composting as a process based on the control of ecologically elective factors. In: Metting, F.B., Jr. (Ed.), Soil Microbial Ecology, Applications in Agricultural and Environmental Management. Marcel Dekker, Inc., New York, 515-544.

Moeskops, B., Sukristiyonubowo, Buchan, D., Sleutel, S., Herawaty, L., Husen, E., Saraswati, R., Setyorini, D. and De Neve, S., 2010. Soil microbial communities and activities under intensive organic and conventional vegetable farming in West Java, Indonesia. *Applied Soil Ecology*, 45, 112-120.

Möller, K., 2015. Effects of anaerobic digestion on soil carbon and nitrogen turnover, N emissions, and soil biological activity. A review. *Agronomy for Sustainable Development*, 35, 1021-1041.

Morse, J. M., 1991. Strategies for sampling, Newbury Park, CA: Sage.

Nevens, F. and Reheul, D., 2003. The application of vegetable, fruit and garden waste (VFG) compost in addition to cattle slurry in a silage maize monoculture: nitrogen availability and use. *European Journal of Agronomy*, 19, 189-203.

Niemeyer, M., Niemeyer, T., Fottner, S., Hardtle, W. and Mohamed, A., 2007. Impact of sod-cutting and choppering on nutrient budgets of dry heathlands. *Biological Conservation*, 134, 344-353.

Nolan, T., Troy, S. M., Healy, M. G., Kwapinski, W., Leahy, J. J. and Lawlor, P. G., 2011. Characterization of compost produced from separated pig manure and a variety of bulking agents at low initial C/N ratios. *Bioresource Technology*, 102, 7131-7138.

Obersteiner, G. and Linzner, R., 2007. Life cycle assessment of composting - a challenge. In: Cossu R., Diaz L.F., Stegmann R. (Eds.), Sardinia 2007 Eleventh International Waste Management and Landfill Symposium, Italy.

OECD, 2013. The Bioeconomy to 2030: designing a policy agenda.

Parkinson, R., Gibbs, P., Burchett, S. and Misselbrook, T., 2004. Effect of turning regime and seasonal weather conditions on nitrogen and phosphorus losses during aerobic composting of cattle manure. *Bioresource Technology*, 91, 171-178.

Parkinson, R., 2000. Enhancing the effective utilisation of animal manures on-farm through compost technology. Final Project Report, MAFF project code WA0519. Ministry of Agriculture, Fisheries and Food, UK.

Patton, M., 1990. Qualitative evaluation and research methods, Sage Publications, Newbury Park, California.

Payá, P., Anastassiades, M., Mack, D., Sigalova, I., Tasdelen, B., Oliva, J. and Barba, A., 2007. Analysis of pesticide residues using the Quick Easy Cheap Effective Rugged and Safe (QuEChERS) pesticide multiresidue method in combination with gas and liquid chromatography and tandem mass spectrometric detection. *Analytical and Bioanalytical Chemistry*, 389, 1697-1714.

Peigne, J. and Girardin, P., 2004. Environmental impacts of farm-scale composting practices. *Water Air and Soil Pollution*, 153, 45-68.

Peltre, C., Thuriès, L., Barthès, B., Brunet, D., Morvan, T., Nicolardot, B., Parnaudeau, V. and Houot, S., 2011. Near infrared reflectance spectroscopy: A tool to characterize the

composition of different types of exogenous organic matter and their behaviour in soil. *Soil Biology and Biochemistry,* 43, 197-205.

Petersen, S.O., Lind, A.M. and Sommer, S.G., 1998. Nitrogen and organic matter losses during storage of cattle and pig manure. *Journal of Agricultural Science*, 130, 69-79.

Pfab, H., Palmer, I., Buegger, F., Fiedler, S., Mueller, T. and Ruser, R., 2011. N2O fluxes from a Haplic Luvisol under intensive production of lettuce and cauliflower as affected by different N-fertilization strategies. *Journal of Plant Nutrition and Soil Science,* 174, 545-553.

Platteau, J., D. Van Gijseghem, and T. Van Bogaert, 2014. Landbouwrapport 2014, Departement Landbouw en Visserij, Brussel.

Powlson, D., Bhogal, A., Chambers, B., Coleman, K., Macdonald, A., Goulding, K. and Whitmore, A., 2012. The potential to increase soil carbon stocks through reduced tillage or organic material additions in England and Wales: A case study. *Agriculture Ecosystems & Environment*, 146, 23-33.

Pronk A., Rijk, B. and Hijbeek, R., 2014. Compatibility of agricultural management practices and types of farming in the EU to enhance climate change mitigation and soil health. Task 4.2 Farm survey country report: The Netherlands.

Rahmani, M., Hodges, A.W. and Kiker, C.F., 2004. Compost Users Attitudes Toward Compost Application In Florida. *Compost Science & Utilization,* 12, 55-60.

Rotz, C.A., 2004. Management to reduce nitrogen losses in animal production. *Journal of Animal Science,* 82(E.Suppl.), E119-E137.

Sánchez-Monedero, M. A., Roig, A., Paredes, C. and Bernal, M. P., 2001. Nitrogen transformation during organic waste composting by the Rutgers system and its effects on pH, EC and maturity of the composting mixtures. *Bioresource Technology*, 78, 301-308.

Schröder, J. J., de Buisonjé, F. E., Kasper, G. J., Verdoes, N. and Verloop, K., 2009. Mestscheiding: relaties tussen techniek, kosten, milieu en landbouwkundige waarde. Plant Research Inernational b.v., Report 287, Wageningen, The Netherlands, 40p.

Seymour, R. M. and Bourdon, M., 2003. Nutrient movement from a windrow of dairy bedding/leaf mulch compost. In: 'Proceedings of the 2003 Georgia Water Resources

Conference'. (Ed. KJ Hatcher). (Institute of Ecology, The University of Georgia: Athens, Georgia).

Shah, G., 2013. Improving the agro-environmental value of solid cattle manure. PhD thesis, Wageningen University, Wageningen, NL. 194p.

Shah, G., Groot, J., Oenema, O. and Lantinga, E., 2012. Covered storage reduces losses and improves crop utilisation of nitrogen from solid cattle manure. *Nutrient Cycling in Agroecosystems*, 94, 299-312.

Shenk, J.S. and Westerhaus, M.O., 1991. Population structuring of near infrared spectra and modified partial least square regression. *Crop Science*, 31, 1548-1555.

Sleutel S., De Neve, S., and Hofman, G., 2007. Assessing causes of recent organic carbon losses from cropland soils by means of regional-scaled input balances for the case of Flanders (Belgium). *Nutrient Cycling in Agroecosystems*. 78, 265-278.

Sleutel S., De Neve, S., Singier, B. and Hofman, G., 2006. Organic C levels in intensively managed arable soils - long-term regional trends and characterization of fractions. *Soil Use Management*, 22, 188-196.

Sleutel, S., De Neve, S., Roibas, M. R. P. and Hofman, G., 2005. The influence of model type and incubation time on the estimation of stable organic carbon in organic materials. *European Journal of Soil Science*, 56, 505-514.

Sleutel S., De Neve, S., Hofman, G., Boeckx, P., Beheydt, D., Van Cleemput, O., Mestdagh, I., Lootens, P., Carlier, L., Van Camp, N., Verbeeck, H., Vande Walle, I., Samson, R., Lust, N. and Lemeur, R., 2003. Carbon stock changes and carbon sequestration potential of Flemish cropland soils. *Global Change Biology*, 9, 1193-1203.

Sommer, S.G. and Dahl, P., 1999. Nutrient and carbon balance during the composting of deep litter. *Journal of Agricultural Engineering Research*, 74, 145-153.

Sorensen, P., 1998. Effects of storage time and straw content of cattle slurry on the mineralization of nitrogen and carbon in soil. *Biology and Fertility of Soils*, 27, 85-91.

Stamatiadis, S., Werner, M. and Buchanan, M., 1999. Field assessment of soil quality as affected by compost and fertilizer application in a broccoli field (San Benito County, California). *Applied Soil Ecology*, 12, 217-225.

Steel, H., Vandecasteele, B., Willekens, K., Sabbe, K., Moens, T. and Bert, W.,2012. Nematode communities and macronutrients in composts and compost-amended soils as affected by feedstock composition. *Applied Soil Ecology*, 61, 100-112.

Stenmarck, Å., Jensen, C., Quested, T. and Moates, G., 2016. Estimates of European food waste levels. IVL Swedish Environmental Research Institute, Stockholm, Sweden, 2016. 80p.

Stinner, W., Möller, K. and Leithold, G., 2008. Effects of biogas digestion of clover/grass-leys, cover crops and crop residues on nitrogen cycle and crop yield in organic stockless farming systems. *European Journal of Agronomy*, 29, 125-134.

Sullivan, D. M., Fransen, S. C., Bary, A. I. and Cogger, C. G., 1998. Fertilizer nitrogen replacement value of food residuals composted with yard trimmings, paper or wood wastes. *Compost Science & Utilization*, 6, 6-18.

Suzuki, T., Ikumi, Y., Okamoto, S. t., Watanabe, I., Fujitake, N. and Otsuka, H., 2004. Aerobic composting of chips from clear-cut trees with various co-materials. *Bioresource Technology*, 95, 121-128.

Terhoeven-Urselmans, T., Schmidt, H., Georg Joergensen, R. and Ludwig, B., 2008. Usefulness of near-infrared spectroscopy to determine biological and chemical soil properties: Importance of sample pre-treatment. *Soil Biology Biochem*istry. 40, 1178-1188.

Tey, Y. S., Li, E., Bruwer, J., Abdullah, A. M., Brindal, M., Radam, A., Ismail, M. M. and Darham, S., 2014. The relative importance of factors influencing the adoption of sustainable agricultural practices: a factor approach for Malaysian vegetable farmers. *Sustainability Science*, 9, 17-29.

The White House, 2012. National Bioeconomy Blueprint. pp. 44.

Thomsen, I. K., Olesen, J. E., Möller, H. and Sorensen, P., 2013. Carbon dynamics and retention in soil after anaerobic digestion of dairy cattle feed and faeces. *Soil Biology and Biochemistry*, 58, 82-87.

Thomsen, I. K. and Olesen, J. E., 2000a. C and N mineralization of composted and anaerobically stored ruminant manure in differently textured soils. *Journal of Agricultural Science*, 135, 151-159.

Thomsen, I. K., 2000b. C and N transformations in 15N cross-labelled solid ruminant manure during anaerobic and aerobic storage. *Bioresource Technology*, 72, 267-274.

Tits, M., Elsen, A., Bries, J. and Vandendriessche, H., 2014. Short-term and long-term effects of vegetable, fruit and garden waste compost applications in an arable crop rotation in Flanders. *Plant and Soil*, 376, 43-59.

Turner, M. S., Clark, G. A., Stanley, C. D. and Smajstrla, A. G., 1994. Physical Characteristics of A Sandy Soil Amended with Municipal Solid-Waste Compost. *Soil and Crop Science Society of Florida Proceedings*, 53, 24-26.

Ulen, B., 1993. Losses of Nutrients Through Leaching and Surface Runoff from Manure-Containing Composts. *Biological Agriculture & Horticulture*, 10, 29-37.

Vance, E.D., Brookes, P.C., Jenkinson, D.S., 1987. An extraction method for measuring soil microbial biomass C. Soil Biology & Biochemistry 19, 703- 707.

Vandekerkhove, K., De Keersmaeker, L., Demolder, H., Esprit, M., Thomaes, A., Van Daele, T. and Van der Aa, B., 2014. Hoofdstuk 13 - Ecosysteemdienst houtproductie (INBO.R.2014. 1993289). In: Stevens, M. et al. (Eds.), Natuurrapport - Toestand en trend van ecosystemen en ecosysteemdiensten in Vlaanderen. Technisch rapport. Mededelingen van het Instituut voor Natuur- en Bosonderzoek, INBO.M.2014. 1988582, Brussel, 75p.

Van Lancker, J., Wauters, E. and Van Huylenbroeck, G., 2016. Managing innovation in the bioeconomy: An open innovation perspective. *Biomass and Bioenergy*, 90, 60-69.

Van Soest, P. J., Robertson, J. B. and Lewis, B. A., 1991. Methods for dietary fiber, neutral detergent fiber, and nonstarch polysaccharides in relation to animal nutrition. *Journal of Dairy Science,* 74, 3583-3597.

Van-Camp, L., Bujarrabal, B., Gentile, A. R., Jones, R., Montanarella, L., Olazabal, C. and Selvaradjou, S-K., 2004. Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection. EUR 21319 EN/3. Office for Official Publications of the European Communities, Luxembourg. 872p.

Vandecasteele, B., Boogaerts, C. and Vandaele, E., 2016. Woody biomass for combustion and green waste composting: balancing green energy and materials recycling? *Waste Management*, submitted.

Vandecasteele, B., Reubens, B., Willekens, K. and De Neve, S., 2014. Composting for increasing the fertilizer value of chicken manure: effects of feedstock on P availability. *Waste Biomass Valor*, 5, 491-503.

Vandecasteele, B., Willekens, K., Zwertvaegher, A., Degrande, L., Tack, F. and Du Laing, G., 2013. Effect of composting on the Cd, Zn and Mn content and fractionation in feedstock mixtures with wood chips from a short-rotation coppice and bark. *Waste Management*, 33, 2195-2203.

Vanden Nest, T., Ruysschaert, G., Vandecasteele, B., Houot, S., Baken, S., Smolders, E., Cougnon, M., Reheul, D. and Merckx, R., 2016. The long term use of farmyard manure and compost: Effects on P availability, orthophosphate sorption strength and P leaching. *Agriculture Ecosystems & Environment*, 216, 23-33.

Vanden Nest, T., 2015. Long term use of different organic fertilizer types and impact on phosphorus leaching, KU Leuven, Belgium.

Vanden Nest, T., Vandecasteele, B., Ruysschaert, G., Cougnon, M., Merckx, R. and Reheul, D., 2014. Effect of organic and mineral fertilizers on soil P and C levels, crop yield and P leaching in a long term trial on a silt loam soil. *Agriculture, Ecosystems & Environment*, 197, 309-317.

Van der Wurff, A.W.G., Fuchs, J.G., Raviv, M., Termorshuizen, A.J. (Eds.), 2016. Handbook for Composting and Compost Use in Organic Horticulture. BioGreenhouse COST Action FA 1105, www.biogreenhouse.org.

Veeken, A., Blok, W., Moolenaar, S. and Grigatti, M., 2007. Standardized determination and classification of compost organic matter stability. In Proceedings of 2nd BOKU Waste Conference, Vienna, Austria, 16-19 April, 2007.

Velthof, G., Lesschen, J., Webb, J., Pietrzak, S., Miatkowski, Z., Pinto, M., Kros, J. and Oenema, O., 2014. The impact of the Nitrates Directive on nitrogen emissions from agriculture in the EU-27 during 2000-2008. *Science of The Total Environment*, 468, 1225-1233.

Viaene, J., Nelissen, V., Vandecasteele, B., Willekens, K., De Neve, S. and Reubens, B., 2016a. Field storage conditions for cattle manure to limit nitrogen losses and optimise fertiliser value. *Animal Production Science*. Advance online publication. doi.org/10.1071/AN16170.

Viaene, J., Van Lancker, J., Vandecasteele, B., Willekens, K., Bijttebier, J., Ruysschaert, G., De Neve, S. and Reubens, B., 2016b. Opportunities and barriers to on-farm composting and compost application: A case study from northwestern Europe. *Waste Management*, 48, 181-192.

Viaene, J., Reubens, B., Vandecasteele, B. and Willekens, K., 2014. Composteren als valorisatievorm van reststromen in de Vlaamse land- en tuinbouw: knelpunten en opportuniteiten. Institute for Agricultural and Fisheries Research (ILVO), Belgium. ILVO-report 167. ISSN 1784-3197.

Vlaamse Landmaatschappij, 2014. Voortgangsrapport Mestbank 2013 over de mestproblematiek in Vlaanderen. 122p.

Vlaco vzw, 2013. Activiteitenverslag 2013. pp. 56.

Walker, P., Williams, D. and Waliczek, T. M., 2006. An analysis of the horticulture industry as a potential value-added market for compost. *Compost Science & Utilization*, 14, 23-31.

Warburton, H. and Sarfo-Mensah, P., 1998. The use of composted urban waste in integrated pest management systems to control pests and pathogens in peri-urban agriculture. Project no. DFID, RNRRS, and NRSP funded program on peri-urban interface (C1045).

Wauters, E. and Mathijs, E., 2013. An investigation into socio-physological determinants of farmers' conservation decisions: method and implications for policy, extension and research. *Journal of Agricultural Education and Extension*, 19, 53-72.

Wauters, E., Bielders, C., Poesen, J., Govers, G. and Mathijs, E., 2010. Adoption of soil conservation practices in Belgium: An examination of the theory of planned behaviour in the agri-environmental domain. *Land Use Policy*, 27, 86-94.

Weber, J., Karczewska, A., Drozd, J., Licznar, M., Licznar, S., Jamroz, E. and Kocowicz, A., 2007. Agricultural and ecological aspects of a sandy soil as affected by the application of municipal solid waste composts. *Soil Biology and Biochemistry*, 39, 1294-1302.

Wichuk, K. M., Tewari, J. P. and McCartney, D.,2011. Plant Pathogen Eradication During Composting: A Literature Review. *Compost Science & Utilization*, 19, 244-266.

Willekens, K., Vandecasteele, B., Buchan, D. and De Neve, S., 2014. Soil quality is positively affected by reduced tillage and compost in an intensive vegetable cropping system. *Applied Soil Ecology*, 82, 61-71.

Willekens, K. and Cloet, B., 2003. Boerderijcompost nieuws. Eerste nieuwsbrief FarmCOMPOST, project van vzw Symbios. TAD FarmCOMPOST, Merelbeke, 8p.

Yahaya, M. S., Kawai, M., Takahashi, J. and Matsuoka, S., 2002. The effect of different moisture contents at ensiling on silo degradation and digestibility of structural carbohydrates of orchardgrass. *Animal Feed Science and Technology*, 101, 127-133.

Zebarth, B. J., Neilsen, G. H., Hogue, E. and Neilsen, D., 1999. Influence of organic waste amendments on selected soil physical and chemical properties. *Canadian Journal of Soil Science*, 79, 501-504.

Zmora-Nahum, S., Markovitch, O., Tarchitzky, J. and Chen, Y. N., 2005. Dissolved organic carbon (DOC) as a parameter of compost maturity. *Soil Biology & Biochemistry*, 37, 2109-2116.

Zucconi, F. and de Bertoldi, M., 1987. Compost specification for the production and characterization of compost from municipal solid waste. In: de Bertoldi, M., Ferranti, M.P., Hermite, P.L., Zucconi, F. (Eds.), Compost: Production, Quality and Use. Elsevier Applied Science Publishers, Barking, 30-50.

Appendix

Table A1 Overview of the questions for farmers during the interviews performed in study 1.

1. Sector? Conventional or organic?		
2. Which byproducts are produced on the farm?		
	· What are the characteristics?	
	· When are they produced?	
	· What are the amounts?	
	· What is the current valorization of the byproducts?	
	· Are there problems in the current valorization option?	
3. Do you use compost?		
YES	· Why?	
	· Amount per ha and per year?	
	· What is the origin (on-farm composted or from an industrial composting company)?	
	INDUSTRIAL	· What is the price?
		· Do you experience problems in the purchase or the quality of compost?
		· Why are you not composting yourself?
	ON-FARM	
		· Do you have specialized composting equipment (machine, monitoring sensors, pile cover)?
		· Do you monitor the process?
		· Where do you compost (on a concrete floor or on the field) and when?
		· Which byproducts do you compost? What are their origin and their amount?
		· What is the amount of compost per year that you make?
		· Are you familiar with the regulation for on-farm composting?
		· Do you experience problems with on-farm composting?
NO	· Why not?	
	· Do you have suggestions to facilitate the use of compost?	

Table A2 Questions asked for each of the identified outcomes, referents and control factors according to the Theory of Planned Behavior in study 2.

		Description	Example	Scale
Outcome	Belief strength	Likelihood of the outcome	Compost improves soil fertility	1: very wrong - 5: very true
	Outcome evaluation	Degree to which the outcome is positively or negatively evaluated by the farmer	Improved soil fertility	1: very bad - 5: very good
Referent	Normative belief	How positive is referent towards compost application	Literature is positive towards compost application	1: strongly disagree - 5: strongly agree
	Motivation to comply	To which degree the farmer values the judgment of the referent	Literature	1: very unimportant - 5: very important
Control	Perceived power	To which degree a factor makes compost application less/more attractive or more difficult/easier	Compost application without the availability of appropriate machinery is	1: very unattractive/difficult - 5: very attractive/easy
	Control belief	To what extent applies each control factor to his farm	The appropriate machinery is not available on my farm	1: very wrong - 5: very true

Table A3 Characterization of feedstock and feedstock mixtures in compost trial 1 and 2 (mean ± standard deviation; n = 4). DM = dry matter.

TRIAL 1

	Crop residues		Composting - Bulking agents				Composting - Feedstock mixtures		Ensiling
	Leek	White cabbage	Poplar wood chips	Wheat straw	Poplar bark	Maize straw	Leek	White cabbage	Maize straw
Fresh bulk density (kg m^{-3})	288	12	195	407	814	192	nda	nda	nda
Organic matter content (% of DM)	16.9 ± 7.0	64.9 ± 3.8	97.5 ± 0.2	93.5 ± 0.2	89.8 ± 0.7	19.4 ± 4.6	53.8 ± 6.6	47.9 ± 3.3	85.5 ± 2.1
Dry matter content (% of fresh weight)	31.8 ± 6.7	11.6 ± 0.7	47.0 ± 0.2	86.1 ± 0.2	50.6 ± 0.7	47.7 ± 2.9	30.3 ± 1.6	39.1 ± 1.0	26.0 ± 4.1
C/N (-)	13.8 ± 2.4	16.4 ± 0.7	103.3 ± 6.3	75.4 ± 4.6	58.3 ± 4.5	33.3 ± 6.4	42.5 ± 3.0	44.2 ± 2.4	76.4 ± 7.8
C/P (-)	51.6 ± 16.3	107.8 ± 6.5	677.6 ± 34.5	554.3 ± 49.6	550.3 ± 92.2	114.4 ± 26.9	nda	nda	500.7 ± 40.2
Total N (g kg^{-1} DM)	6.67 ± 1.85	21.96 ± 1.26	5.26 ± 0.31	6.91 ± 0.45	8.60 ± 0.69	3.22 ± 0.34	7.07 ± 1.02	6.02 ± 0.10	6.26 ± 0.53
Total P (g kg^{-1} DM)	1.81 ± 0.24	3.37 ± 0.11	0.81 ± 0.00	0.95 ± 0.09	0.93 ± 0.15	0.95 ± 0.05	nda	nda	0.95 ± 0.06
Biodegradation potential (-)	nda	nda	nda	nda	nda	nda	3.4 ± 0.4	2.1 ± 0.3	nda

TRIAL 2

	Crop residues		Composting - Bulking agents				Composting - Feedstock mixtures		Ensiling
	Leek	Leek$_{extra}$	Rye straw	Poplar bark	Chopped heath	Strawberry substrate	Leek$_{heath}$	Leek$_{strawberry}$	Maize straw
Fresh bulk density (kg m^{-3})	332 ±161	186 ± 18	12	278 ± 19	266	374	213 ± 51	322 ± 52	nda
Organic matter content (% of DM)	54.3 ± 11.7	57.4 ± 11.5	96.1 ± 0.4	88.7 ± 0.2	72.6 ± 4.3	88.2 ± 0.9	65.6 ± 10.8	74.4 ± 9.0	91.6 ± 0.3
Dry matter content (% of fresh weight)	13.2 ± 1.9	13.0 ± 2.3	64.0 ± 12.4	37.3 ± 2.8	35.3 ± 0.1	19.2 ± 0.4	26.9 ± 3.1	18.3 ± 1.9	18.3 ± 0.3
C/N (-)	15.6 ± 2.7	17.4 ± 1.2	118.4 ± 10.1	51.1 ± 5.2	39.3 ± 6.5	35.5 ± 1.6	32.8 ± 5.4	27.7 ± 4.5	50.2 ± 0.5
C/P (-)	110.1 ± 14.2	117.6 ± 9.4	465.4 ± 140.8	526.9 ± 64.5	798.5 ± 67.9	270.8 ± 8.1	nda	nda	608.4 ± 35.8
Total N (g kg^{-1} DM)	19.39 ± 3.34	18.50 ± 4.26	4.54 ± 0.36	9.72 ± 0.97	10.40 ± 1.08	13.82 ± 0.64	11.15 ± 0.88	15.03 ± 0.70	10.14 ± 0.07
Total P (g kg^{-1} DM)	2.76 ± 0.59	2.72 ± 0.58	1.22 ± 0.33	0.95 ± 0.13	0.51 ± 0.03	1.81 ± 0.04	nda	nda	0.84 ± 0.05
Biodegradation potential (-)	7.9 ± 3.6	8.0 ± 3.5	nda	nda	nda	nda	3.0 ± 1.0	2.5 ± 0.6	nda

Table A4 Physico-chemical quality of the compost mixtures at day 0, 14, 27, 55 and 76 of co-composting leek residues with wood chips (Cw), chopped heath biomass (Ch), spent strawberry (Cs) or tomato (Ct) substrate (mean ± standard deviation, $n = 4$). DM = dry matter, OM = organic matter, nda = no data available. Parameters with the same letter in the same sampling time are not significantly different between the treatments (Scheffé test, $p < 0.05$).

	Cw					Ch					Cs					Ct				
	Day 1	Day 14	Day 27	Day 55	Day 76	Day 1	Day 14	Day 27	Day 55	Day 76	Day 1	Day 14	Day 27	Day 55	Day 76	Day 1	Day 14	Day 27	Day 55	Day 76
Fresh bulk density (kg m^{-3})	270 a	291 a	370 a	463 a	470 a	213 a	388 b	453 b	505 bc	514 b	322 a	405 b	508 c	527 c	492 ab	250 a	387 b	487 c	497 b	476 ab
Organic matter content (% of DM)	80.5 a	83.1 c	77.6 c	77.0 c	61.7 bc	65.6 a	56.9 a	53.3 a	54.3 a	47.9 a	74.4 a	78.9 c	73.4 c	72.6 bc	71.4 c	66.6 a	69.7 b	63.8 b	68.3 b	61.5 b
Dry matter content (% of fresh weight)	22.5 ab	24.8 c	22.7 b	26.5 b	28.2 b	26.9 b	30.0 d	28.4 c	30.5 c	31.4 c	18.3 a	20.0 a	19.9 a	21.7 a	22.8 a	24.8 ab	23.4 b	22.5 b	25.3 b	25.3 b
C/N (-)	40.7 a	54.9 b	49.6 c	43.4 b	25.2 b	32.8 a	27.2 a	23.7 a	28.9 a	21.6 a	27.7 a	32.2 a	29.1 b	30.4 a	28.2 b	25.7 a	31.3 a	29.5 b	33.7 a	28.4 b
C/P (-)	nda	348 c	318 c	315 c	157 b	nda	335 c	303 c	336 c	232 d	nda	217 b	193 b	195 b	173 c	nda	72 a	58 a	82 a	60 a
Total N (g kg^{-1} DM)	11.65 a	8.41 a	8.70 a	8.89 a	13.68 a	11.15 ab	11.63 b	12.52 b	10.43 ab	12.37 a	15.03 b	13.61 c	14.01 c	13.30 c	14.08 a	14.46 ab	12.35 b	12.04 b	11.27 b	12.05 a
Total P (g kg^{-1} DM)	nda	1.34 a	1.36 b	1.39 a	2.19 b	nda	0.95 a	0.98 a	0.91 a	1.14 a	nda	2.02 b	2.11 c	2.09 b	2.29 b	nda	5.39 c	6.13 d	4.65 c	5.69 c
pH-H_2O (-)	nda	7.22 b	7.34 a	7.87 b	8.69 d	nda	6.63 a	7.44 a	7.27 a	7.41 a	nda	7.30 b	7.84 b	7.82 b	7.79 b	nda	7.18 b	7.88 b	7.94 c	7.92 c
Electrical conductivity (µS cm^{-1})	nda	300 b	335 a	336 b	397 c	nda	243 a	304 a	238 a	247 a	nda	368 c	432 b	332 b	340 b	nda	590 d	540 c	435 c	436 d
NO_3^--N (mg L^{-1})	nda	< 5	5	< 5	< 5	nda	< 5	3	< 5	< 5	nda	< 5	< 5	8	17	nda	72	7	5	12
NH_4^+-N (mg L^{-1})	nda	< 5	< 5	< 5	< 5	nda	49	43	< 5	< 5	nda	40	12	< 5	< 5	nda	32	10	< 5	< 5
SO_4^{2-} (mg L^{-1})	nda	31 a	53 bc	44 b	45 b	nda	< 11.7 a	12 a	< 11.7 a	14 a	nda	123 b	52 b	41 b	104 c	nda	356 c	212 c	145 c	255 d
Cl^- (mg L^{-1})	nda	91 a	101 a	126 a	147 ab	nda	101 a	135 b	114 a	114 a	nda	120 a	145 b	78 a	138 a	nda	190 b	218 c	140 a	208 b
Oxygen uptake rate (mmol kg^{-1} OM h^{-1})	nda	28.0 b	25.6 c	11.3 b	7.3 b	nda	8.4 a	8.3 a	4.6 a	3.3 ab	nda	9.7 a	18.9 b	4.5 a	2.7 a	nda	8.7 a	6.6 a	3.1 a	2.9 a
Hemicellulose (% of OM)	20.3 ab	22.2 b	21.6 b	21.7 b	22.8 b	23.4 b	17.7 a	15.8 a	17.8 b	17.4 a	17.4 a	16.5 a	12.8 a	12.9 a	15.5 a	20.2 ab	16.5 a	13.7 a	13.5 a	17.9 a
Cellulose (% of OM)	43.1 a	46.4 c	45.2 c	35.5 b	32.5 b	42.5 a	28.2 a	23.7 a	28.0 a	24.1 a	37.1 a	35.0 b	29.6 b	33.1 ab	29.3 b	40.1 a	34.3 b	32.0 b	31.7 ab	31.1 b
Lignin (% of OM)	13.3 a	18.3 a	19.8 a	25.6 a	31.3 a	24.4 a	34.2 b	35.6 b	39.1 c	40.3 b	22.2 a	32.9 b	34.3 b	34.2 b	32.9 ab	24.1 a	32.3 b	35.5 b	33.8 b	32.0 a
Biodegradation potential (-)	4.9 b	3.8 b	3.4 b	2.2 c	1.8 c	3.0 a	1.3 a	1.1 a	1.2 a	1.0 a	2.5 a	1.6 b	1.2 a	1.3 b	1.4 b	2.5 a	1.6 b	1.3 a	1.3 ab	1.5 bc

Table A5 Estimated coefficients and standard errors (s.e.) of the linear regression model: $Y = \beta_0 + \beta_{1,i} * \text{Treatment} + \beta_2 * \text{Day} + \beta_{3,i} * \text{Day} * \text{Treatment} + \epsilon$, for the different compost characteristics Y. With Treatment 1 = co-composted leek residues with wood chips (Cw), treatment 2 = co-composted leek residues with chopped heath biomass (Ch), treatment 3 = co-composted leek residues with spent strawberry substrate (Cs) and treatment 4 = co-composted leek residues with spent tomato substrate (Ct). $p < 0.05$ indicates a significant effect.

	β_0 (s.e.)	*p*	$\beta_{1,2}$ (s.e.)	*p*	$\beta_{1,3}$ (s.e.)	*p*	$\beta_{1,4}$ (s.e.)	*p*	β_2 (s.e.)	*p*	$\beta_{3,2}$ (s.e.)	*p*	$\beta_{3,3}$ (s.e.)	*p*	$\beta_{3,4}$ (s.e.)	*p*
Organic matter content (% of DM)	84.1 (1.8)	<0.01	-22.1 (2.5)	<0.01	-7.7 (2.5)	<0.01	-16.1 (2.5)	<0.01	-0.2 (0.0)	<0.01	0.05 (0.06)	0.38	0.2 (0.1)	<0.01	0.2 (0.1)	<0.01
Total N (g kg^{-1} DM)	9.2 (0.5)	<0.01	2.3 (0.7)	<0.01	5.2 (0.7)	<0.01	4.2 (0.7)	<0.01	0.04 (0.01)	<0.01	-0.03 (0.02)	0.04	-0.04 (0.02)	<0.01	-0.06 (0.02)	<0.01
Dry matter content (% of fresh weight)	22.4 (0.6)	<0.01	5.3 (0.8)	<0.01	-3.8 (0.8)	<0.01	1.1 (0.8)	0.16	0.07 (0.01)	<0.01	0.02 (0.02)	0.21	-0.02 (0.02)	0.37	-0.05 (0.02)	<0.01
Fresh bulk density (kg m^{-3})	271 (21)	<0.01	24 (30)	0.43	104 (30)	<0.01	58 (30)	0.06	2.9 (0.5)	<0.01	0.5 (0.7)	0.43	-0.8 (0.7)	0.26	-0.3 (0.7)	0.63
C/N (-)	51.7 (2.1)	<0.01	-21.5 (2.9)	<0.01	-21.9 (2.9)	<0.01	-23.2 (2.9)	<0.01	-0.3 (0.1)	<0.01	0.2 (0.1)	0.02	0.3 (0.1)	<0.01	0.3 (0.1)	<0.01
Hemicellulose (% of OM)	20.9 (1.0)	<0.01	-0.1 (1.4)	0.57	-4.8 (1.4)	<0.01	-3.6 (1.4)	0.01	0.02 (0.02)	0.31	-0.07 (0.03)	0.02	-0.1 (0.03)	0.09	-0.1 (0.03)	0.11
Cellulose (% of OM)	46.8 (1.5)	<0.01	-11.8 (2.1)	<0.01	-11.1 (2.1)	<0.01	-9.6 (2.1)	<0.01	-0.2 (0.03)	<0.01	0.02 (0.05)	0.75	0.1 (0.05)	0.04	0.1 (0.05)	0.08
Lignin (% of OM)	13.8 (1.5)	<0.01	14.7 (2.1)	<0.01	14.0 (2.10)	<0.01	15.2 (2.1)	<0.01	0.2 (0.03)	<0.01	-0.05 (0.05)	0.33	-0.1 (0.05)	<0.01	-0.2 (0.05)	<0.01
Biodegradation potential (-)	4.6 (0.2)	<0.01	-2.4 (0.3)	<0.01	-2.6 (0.3)	<0.01	-2.6 (0.3)	<0.01	-0.04 (<0.01)	<0.01	0.02 (0.01)	<0.01	0.03 (0.01)	<0.01	0.03 (0.01)	<0.01
C/P (-)	399 (17)	<0.01	-44 (24)	0.07	179 (24)	<0.01	-330 (24)	<0.01	-2.8 (0.4)	<0.01	1.5 (0.5)	<0.01	2.2 (0.5)	<0.01	2.7 (0.5)	<0.01
Total P (mg kg^{-1} DM)	1041 (197)	<0.01	-141 (278)	0.61	934 (278)	<0.01	4624 (278)	<0.01	10.8 (3.99)	<0.01	-8.51 (5.64)	0.14	-7.22 (5.64)	0.21	-15.43 (5.64)	<0.01
pH-H_2O (-)	6.77 (0.11)	<0.01	0.03 (0.16)	0.83	0.66 (0.16)	<0.01	0.55 (0.16)	<0.01	0.02 (<0.01)	<0.01	-0.01 (<0.01)	<0.01	-0.02 (<0.01)	<0.01	-0.01 (<0.01)	<0.01
Electrical conductivity (µS cm^{-1})	285 (19)	<0.01	-10 (27)	0.69	127 (27)	<0.01	328 (27)	<0.01	1.3 (0.4)	<0.01	-1.7 (0.5)	<0.01	-2.3 (0.5)	<0.01	-4.0 (0.5)	<0.01
NO_3^--N (mg L^{-1})	6.80 (5.69)	0.24	-0.87 (8.05)	0.91	-5.87 (8.05)	0.47	51.02 (8.05)	<0.01	0.02 (0.12)	0.85	0.01 (0.16)	0.95	0.20 (0.16)	0.23	-0.73 (0.16)	<0.01
NH_4^+-N (mg L^{-1})	5.00 (3.59)	0.17	55.66 (5.07)	<0.01	31.65 (5.07)	<0.01	24.08 (5.07)	<0.01	<0.01	1.00	-0.81 (0.10)	<0.01	-0.49 (0.10)	<0.01	-0.37 (0.10)	<0.01
SO_4^{2-} (mg L^{-1})	39 (23)	0.10	-27 (33)	0.41	50 (33)	0.13	269 (33)	<0.01	0.11 (0.47)	0.82	-0.08 (0.67)	0.9	-0.31 (0.67)	0.64	-1.64 (0.67)	0,02
Cl^- (mg L^{-1})	78 (16)	<0.01	38 (23)	0.10	48 (23)	0.04	120 (23)	<0.01	0.9 (0.3)	<0.01	-0.9 (0.5)	0.06	-1.0 (0.5)	0.03	-1.1 (0.5)	0.02
Oxygen uptake rate (mmol kg^{-1} OM h^{-1})	33.6 (2.6)	<0.01	-23.6 (3.6)	<0.01	-16.5 (3.6)	<0.01	-24.2 (3.6)	<0.01	-0.4 (0.1)	<0.01	0.3 (0.1)	<0.01	0.2 (0.1)	0.03	0.3 (0.1)	<0.01

Table A6 Pesticide residues in $LC2_{strawberry}$ (mg kg^{-1} compost).

Dithiocarbamaat	< report limit
Cyhalothrin (sum of gamma and lambda)	0.020
Fludioxonil	0.25
Endosulfan (sum of alpha- and beta-isomers and endosulfansulphate expressed as endosulfan)	0.033
Boscalid	1.1
Clofentezine	0.022
Dimethomorph (sum of isomers)	0.44
Fenhexamid	0.034
Hexythiazox	0.011
Penconazole	0.073
Quinoxyfen	0.043
Tebufenpyrad	0.041
Tetraconazole	0.18
Spinosad (sum of spinosyn A and D, expressed as spinosad)	0.040

Table A7 Soil characterization of the 0-30 cm layer (mean ± standard deviation; $n = 3$) at the start of Experiment 2. Parameters with the same letter are not significantly different from each other (Scheffé test, $p < 0.05$).

	Wachtebeke		Zoersel		Zwevezele	
Dry bulk density (g cm^{-3})	1.29 ± 0.05	ab	1.56 ± 0.04	a	1.50 ± 0.05	b
Water content (% of dry soil)	13.6 ± 0.7	a	14.3 ± 0.3	a	17.4 ± 0.2	b
Organic carbon (% of dry soil)	1.55 ± 0.40	a	1.45 ± 0.34	a	0.91 ± 0.01	a
pH-KCl (-)	4.66 ± 0.06	b	4.94 ± 0.02	a	4.53 ± 0.03	c
N (% of dry soil)	0.13 ± 0.04	a	0.11 ± 0.04	a	0.08 ± 0.00	a

Table A8 Product quality of the cattle farmyard manure in Experiment 2 at the start (mean ± standard deviation; *n* = 4) and after storage without handling (S2), storage with a TopTex cover (ST2), storage with a plastic cover (SP2), and composting in combination with covering with TopTex cover (C2) (n = 3). DM = dry matter, FM = fresh matter and OM = organic matter.

	Wachtebeke					Zoersel					Zwevezele				
	Day 1	Day 61				Day 1	Day 64				Day 1	Day 54			
		S2	ST2	SP2	C2		S2	ST2	SP2	C2		S2	ST2	SP2	C2
Fresh bulk density (kg m^{-3})	347 ± 32	513 ± 68	485 ± 20	398 ± 31	507 ± 45	599 ± 45	652 ± 21	634 ± 6	592 ± 36	642 ± 26	656 ± 13	757 ± 2	715 ± 22	675 ± 26	718 ± 13
pH-H_2O (-)	8.5 ± 0.1	8.9 ± 0.1	9.0 ± 0.2	9.0 ± 0.0	9.1 ± 0.1	8.8 ± 0.2	8.5 ± 0.2	8.7 ± 0.1	8.5 ± 0.1	8.8 ± 0.1	8.7 ± 0.1	8.8 ± 0.2	8.4 ± 0.2	8.5 ± 0.1	8.7 ± 0.2
Electrical conductivity (µS cm^{-1})	1491 ± 176	1619 ± 469	2160 ± 367	1646 ± 200	1915 ± 229	2593 ± 277	1833 ± 129	2587 ± 223	2527 ± 57	2897 ± 386	2313 ± 171	2797 ± 301	2513 ± 84	2397 ± 153	2423 ± 284
Organic matter content (% of DM)	80.1 ± 1.4	73.9 ± 4.0	53.7 ± 9.2	75.6 ± 0.4	50.6 ± 4.0	75.7 ± 2.3	59.5 ± 7.3	66.1 ± 5.6	69.6 ± 3.9	57.9 ± 1.4	73.2 ± 7.4	65.9 ± 9.0	75.7 ± 2.7	72.1 ± 1.4	66.6 ± 4.1
Dry matter content (% of FM)	22.8 ± 0.9	20.2 ± 1.2	25.4 ± 1.0	21.2 ± 1.8	29.7 ± 5.1	22.3 ± 2.3	22.6 ± 2.2	24.6 ± 1.5	23.3 ± 0.5	25.5 ± 2.6	19.9 ± 0.5	21.9 ± 1.1	19.1 ± 0.2	19.2 ± 0.2	20.2 ± 0.5
Volumetric moisture content (kg m^{-3} FM)	268 ± 27	409 ± 54	362 ± 19	314 ± 29	356 ± 24	450 ± 49	505 ± 30	478 ± 9	454 ± 29	479 ± 36	526 ± 14	591 ± 7	579 ± 19	545 ± 22	573 ± 13
NO_3^--N (mg kg^{-1} DM)	2 ± 1	231 ± 298	1170 ± 518	51 ± 78	839 ± 500	1 ± 0	8 ± 6	12 ± 13	1 ± 0	9 ± 10	1 ± 0	7 ± 2	3 ± 2	6 ± 5	12 ± 8
NH_4^+-N (mg kg^{-1} DM)	2727 ± 939	2009 ± 438	1431 ± 687	2003 ± 352	491 ± 554	3283 ± 1727	958 ± 190	1378 ± 667	1322 ± 369	1860 ± 543	4018 ± 1435	3339 ± 340	3093 ± 547	3784 ± 363	3271 ± 297
NO_3^--N / NH_4^+-N (-)	0.0007 ± 0.0005	0.144 ± 0.188	0.900 ± 0.588	0.031 ± 0.047	3.669 ± 4.680	0.0003 ± 0.0001	0.008 ± 0.006	0.008 ± 0.010	0.001 ± 0.000	0.005 ± 0.006	0.0002 ± 0.0001	0.002 ± 0.001	0.001 ± 0.001	0.001 ± 0.001	0.004 ± 0.002
N (g kg^{-1} DM)	26.1 ± 2.0	28.1 ± 3.4	26.2 ± 6.2	25.7 ± 3.6	26.0 ± 3.7	33.0 ± 5.7	25.5 ± 3.3	27.8 ± 1.8	29.8 ± 4.1	28.2 ± 2.2	27.3 ± 2.0	26.1 ± 2.2	27.3 ± 0.6	27.6 ± 1.8	25.9 ± 0.8
P (g kg^{-1} DM)	3.9 ± 0.7	5.7 ± 0	5.1 ± 0	5.4 ± 0	5.8 ± 0	6.9 ± 1.0	5.6 ± 0	7.3 ± 0	7.5 ± 0	8.7 ± 0	5.7 ± 0.6	5.9 ± 0	5.6 ± 0	5.9 ± 0	5.8 ± 0
C/N (-)	17.1 ± 1.1	14.7 ± 1.0	11.5 ± 1.1	16.5 ± 2.3	10.9 ± 0.7	13.0 ± 2.0	13.0 ± 1.0	13.3 ± 1.6	13.1 ± 1.3	11.5 ± 1.0	14.9 ± 0.7	14.0 ± 1.1	15.4 ± 0.7	14.6 ± 1.2	14.3 ± 0.9
C/P (-)	118 ± 21	72 ± 5	60 ± 8	82 ± 20	49 ± 3	62 ± 7	60 ± 2	51 ± 6	54 ± 12	37 ± 1	72 ± 4	62 ± 8	75 ± 4	67 ± 2	63 ± 2
K (g kg^{-1} DM)	27.7 ± 1.8	30.2 ± 3.9	34.8 ± 5.7	31.1 ± 1.5	36.1 ± 2.7	32.5 ± 1.7	24.3 ± 1.6	34.4 ± 3.0	34.6 ± 1.9	38.2 ± 1.5	32.4 ± 1.7	25.9 ± 4.2	32.3 ± 1.5	30.8 ± 2.9	31.1 ± 1.2
Ca (g kg^{-1} DM)	9.0 ± 1.5	13.7 ± 1.8	14.3 ± 4.6	11.6 ± 2.0	15.4 ± 1.0	7.6 ± 1.2	6.7 ± 1.2	8.7 ± 1.0	10.0 ± 2.7	14.8 ± 0.8	11.2 ± 1.1	11.8 ± 1.1	10.5 ± 0.2	10.8 ± 0.8	10.9 ± 0.1
Oxygen Uptake Rate (mmol kg^{-1} OM h^{-1})	31.2 ± 5.0	17.7 ± 6.0	10.5 ± 11.0	24.7 ± 5.7	15.6 ± 0.5	39.3 ± 5.4	35.1 ± 3.6	18.6 ± 6.1	23.5 ± 8.5	13.9 ± 2.0	43.0 ± 13.9	19.8 ± 9.1	32.4 ± 4.7	29.6 ± 8.1	20.2 ± 4.8
Biodegradation potential (-)	7.8 ± 0.9	3.6 ± 0.2	2.6 ± 0.6	4.2 ± 0.5	2.0 ± 0.6	5.4 ± 0.5	4.6 ± 0.5	4.1 ± 0.3	4.2 ± 0.6	2.4 ± 0.4	4.2 ± 0.5	3.2 ± 0.2	4.1 ± 0.5	4.1 ± 0.3	3.6 ± 0.4

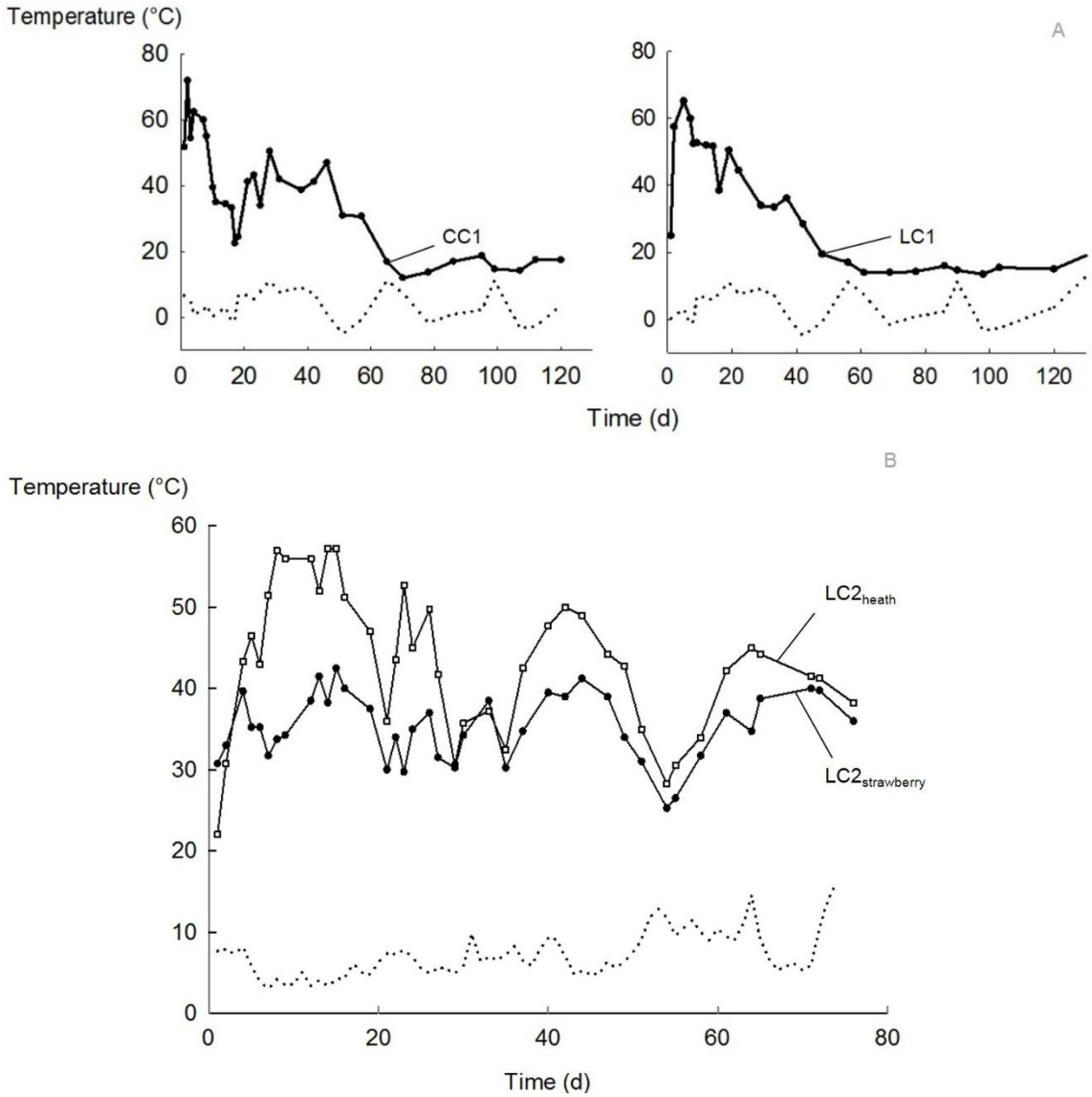
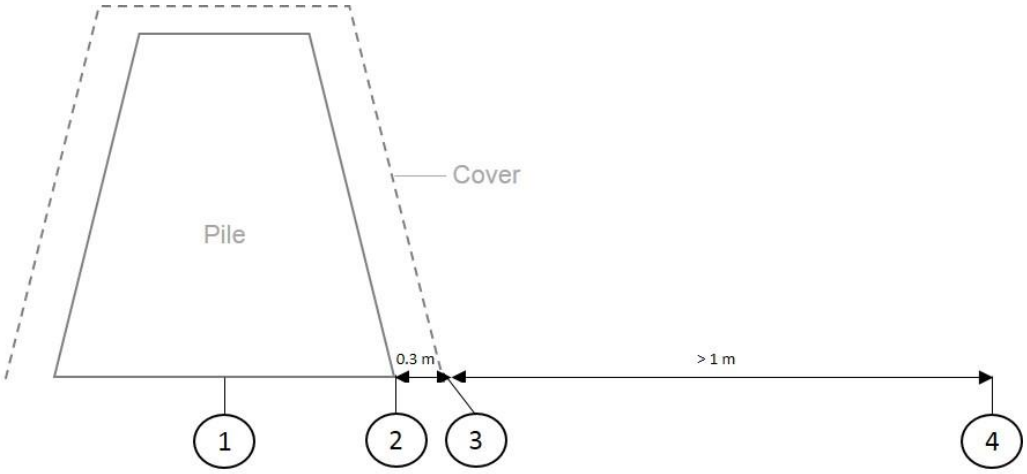


Figure A1 Temperature profiles of the compost piles from trial 1 (A), with CC1 = composted cabbage residues and LC1 = composted leek residues; and trial 2 (B), with $LC2_{heath}$ = composted leek residues with heath chopper and $LC2_{strawberry}$ composted leek residues with strawberry substrate (mean of 4 measurements, standard deviations were below 10°C). Ambient air temperature is represented by the dotted lines.



1. Samples under the pile
2. Samples at the border of the pile, under the cover
3. Samples at the border of the pile, just outside the cover (at 30 cm)
4. Reference samples, at > 1 m from the pile

Figure A2 Position of the soil samples in Experiment 1 and 2.

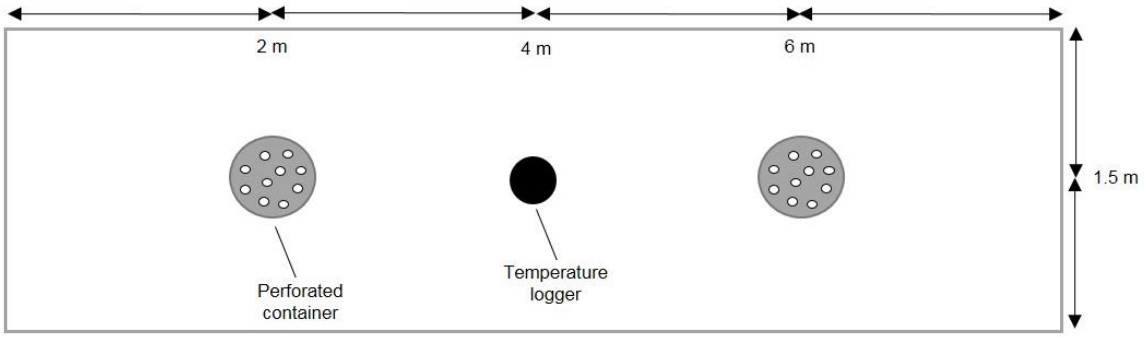


Figure A3 Position of the containers and the temperature logger under the piles in Experiment 2.

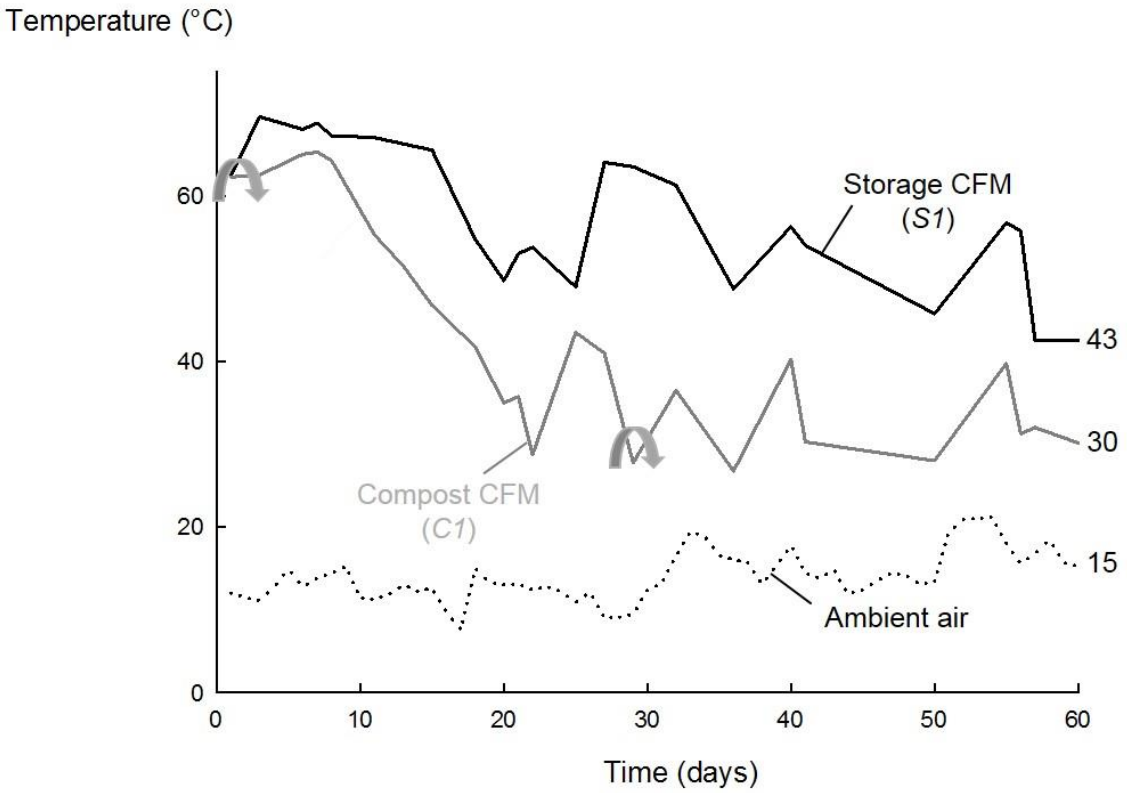


Figure A4 Temperature profiles of the stored and composted cattle farmyard manure (CFM) in Experiment 1 (mean of 4 measurements, standard deviations were below 10°C). The arrows indicate when the pile was turned with a compost turner.

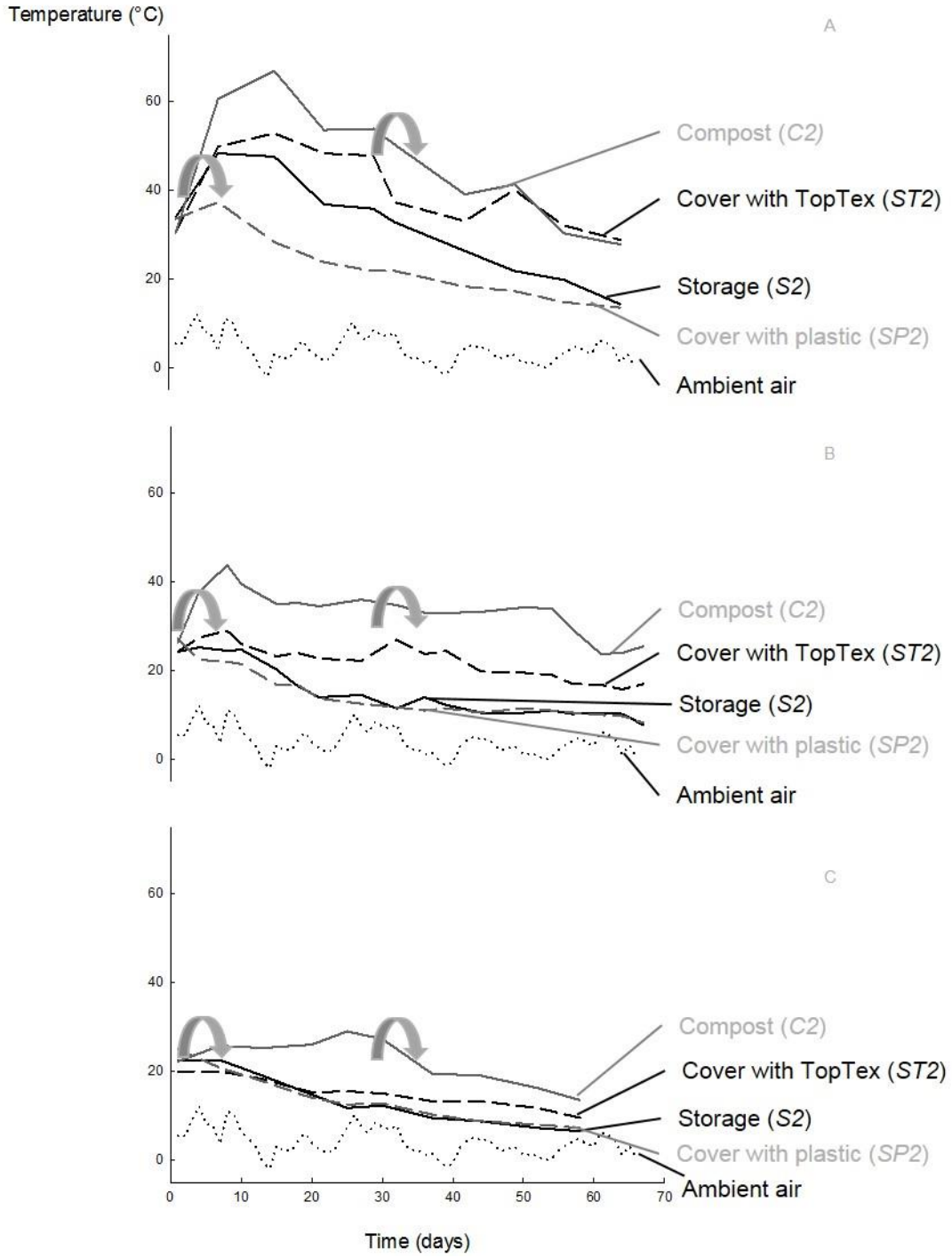


Figure A5 Temperature profiles of the stored (S2), covered with plastic (SP2), covered with TopTex (ST2) and composted (C2) cattle farmyard manure in Wachtebeke (A), Zoersel (B) and Zwevezele (C) (mean of 4 measurements, standard deviations were below 10°C) in Experiment 2. The arrows indicate when the composted piles were turned with a compost turner.

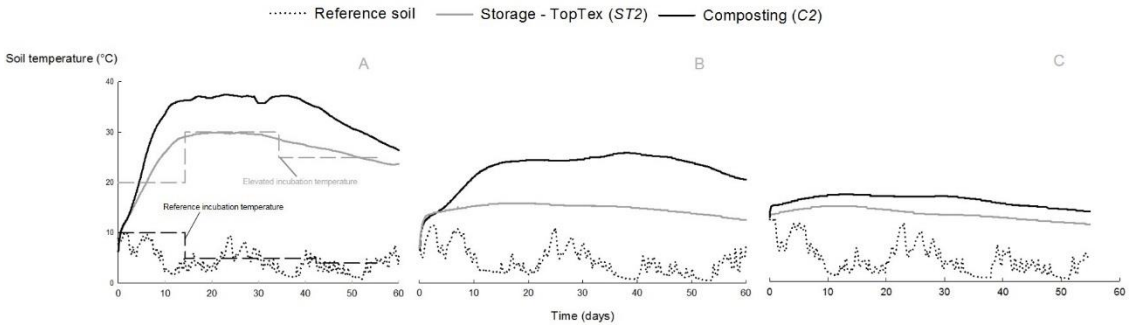


Figure A6 Soil temperature profiles at 10 cm depth in a reference situation (soil not covered by a manure pile) and under the covered with TopTex and composted cattle farmyard manure in Wachtebeke (A), Zoersel (B) and Zwevezele (C) in Experiment 2. The simulated reference and elevated soil temperatures in the incubation experiment are shown in Figure A.

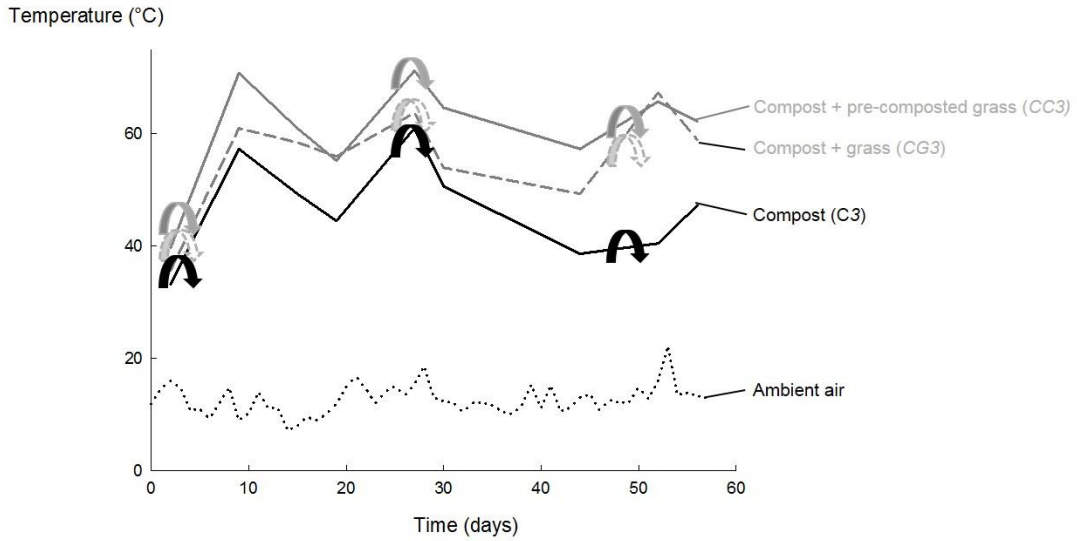


Figure A7 Temperature profiles of cattle farmyard manure composted (C3), co-composted with hay of grass (CG3) and co-composted with pre-composted hay of grass (CC3) (mean of 4 measurements, standard deviations were below 10°C) in Experiment 3. The arrows indicate when the piles were turned with a compost turner.

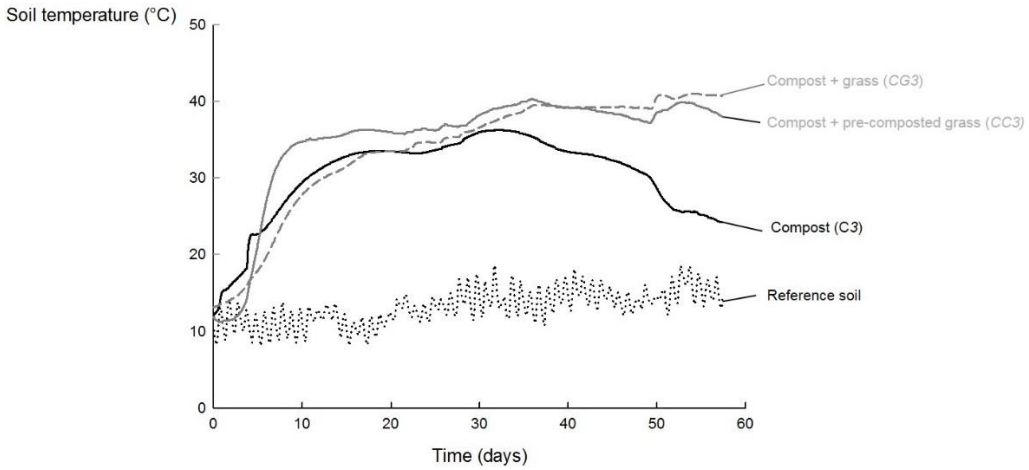


Figure A8 Soil temperature profiles at 10 cm depth in a reference situation (soil not covered by a manure pile) and under the cattle farmyard manure that was composted (C3), co-composted with hay of grass (CG3) and co-composted with pre-composted hay of grass (CC3) in Experiment 3.

Curriculum Vitae

Personal information

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2012-2016: PhD in Bioscience Engineering

PhD-research: 'Optimal valorization of organic-biological byproducts from the primary sector with a focus on composting'

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2010-2012: Master of Bioscience Engineering, Land and Forest Management

Master-thesis: 'Bud drop after dark transport in *Phalaenopsis* orchids' - Supervisors: Prof. Dr. ir. Kathy Steppe and Dr. ir. Veerle De Schepper

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2007-2010: Bachelor of Bioscience Engineering, Land and Forest Management

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Scientific publications in journals with peer review

Viaene, J., Agneessens, L., Capito, C., Ameloot, N., Reubens, B., Willekens, K., Vandecasteele, B. and De Neve, S., 2016. Co-ensiling, co-composting and anaerobic co-digestion of vegetable crop residues: Product stability and effect on soil carbon and nitrogen dynamics. Manuscript submitted for publication.

Viaene, J., Nelissen, V., Reubens, B., Willekens, K., Driehuis, F., De Neve, S. and Vandecasteele, B., 2016. Improving the product stability and fertilizer value of cattle slurry solid fraction through co-composting or co-ensiling. Manuscript submitted for publication.

Viaene, J., Nelissen, V., Vandecasteele, B., Willekens, K., De Neve, S. and Reubens, B., 2016. Field storage conditions for cattle manure to limit nitrogen losses and optimize fertilizer value. *Animal Production Science*. Advance online publication. doi.org/10.1071/AN16170.

Viaene, J., Reubens, B., Willekens, K., Van Waes, C., De Neve, S. and Vandecasteele, B., 2016. Potential of chopped heath biomass and spent growth media to replace wood chips as bulking agent for composting vegetable crop residues. Manuscript submitted for publication.

Viaene, J., Van Lancker, J., Vandecasteele, B., Willekens, K., Bijttebier, J., Ruysschaert, G., De Neve, S. and Reubens, B., 2016. Opportunities and barriers to on-farm composting and compost application: A case study from northwestern Europe. *Waste Management*, 48, 181-192.

Agneessens, L., **Viaene, J.**, Vanden Nest, T., Vandecasteele, B. and De Neve, S., 2015. Effect of ensilaged vegetable crop residue amendments on soil carbon and nitrogen dynamics. *Scientia Horticulturae*, 192, 311-319.

Conferences and symposia

International conferences with oral presentation

Viaene, J., Nelissen, V., Vandecasteele, B., Willekens, K., De Neve, S. and Reubens, B., 2015. Storage conditions for cattle farmyard manure on the field: limiting nutrient losses and optimizing fertilizer value. ManuREsource 2015, 2nd international conference on manure management and valorization, Ghent, Belgium, 4/12/2015.

Viaene, J., Nelissen, V., Vandecasteele, B., Willekens, K., De Neve, S. and Reubens, B., 2015. Improving product quality and limiting nutrient losses of solid fraction of cattle slurry by composting and ensiling. RAMIRAN 2015, Hamburg, Germany, 9/09/2015.

Viaene, J., Van Lancker, J., Reubens, B., Vandecasteele, B. and Willekens, K., 2014. Composting: the solution to closing cycles and improving soil quality in the transition towards a bio-economy? ISEE conference Iceland 2014, Reykjavik, Iceland, 13/08/2014.

International conferences with poster presentation

Viaene, J., Nelissen, V., Reubens, B., Willekens, K., De Neve, S. and Vandecasteele, B., 2015. Composting, co-composting or co-ensiling the solid fraction of cattle slurry to improve product quality and limit nutrient losses. ManuREsource 2015: 2nd international conference on manure management and valorization, Ghent, Belgium, 4/12/2015.

Viaene, J., Nelissen, V., Vandecasteele, B., Willekens, K. and De Neve, S., 2015. Limiting nutrient losses and improving product quality during storage of cattle manure by composting and ensiling. RAMIRAN 2015, Hamburg, Germany, 9/09/2015.

Vandecasteele, B., Viaene, J., Agneessens, L., Van De Sande, T., Dekeyser, D. and De Neve, S., 2014. Vegetable crop residues as feedstock for composting and silage: collection efficiency and product quality. ORBIT Conference 2014, Gödöllő, Hungary, 26/06/2014.

Van Lancker, J., Mondelaers, K., Viaene, J., Kips, L., Hanseeuw, E. and Van Huylenbroeck, G., 2014. Supporting system innovation in the bio-economy: illustration of an

innovation feasibility evaluation instrument. ISEE conference Iceland 2014, Reykjavik, Iceland, 13/08/2014.

Viaene, J., Reubens, B., Vandecasteele, B., Willekens, K. and De Neve, S., 2013. Removing vegetable crop residues for composting and ensilaging: effects on N, P and organic matter. Nutrihort Conference: Nutrient management, innovative techniques and nutrient legislation in intensive horticulture for an improved water quality, Ghent, Belgium, 16/09/2013.

Viaene, J., Reubens, B., Vandecasteele, B., Willekens, K. and De Neve, S., 2013. Use of by-products as system innovation: GeNeSys: Closing cycles trough composting. 9th International Conference on Renewable Resources & Biorefineries, Antwerp, Belgium, 5/06/2013.

Viaene, J., Reubens, B., Vandecasteele, B., Willekens, K. and De Neve, S., 2013. Valorisation of by-products from agriculture and nature management: Opportunities for potting media, composting or as soil improver. RAMIRAN 2013, Versailles, France, 3/06/2013.

National symposia and workshops with oral or poster presentation

Viaene, J., Nelissen, V., Willekens, K., Vandecasteele, B. and Reubens, B., 2015. Stalmestopslag op de kopakker: Hoe risico op uitspoeling beperken en een waardevol product maken? CriNglooP Collectief, Merelbeke, Belgium, 8/10/2015. Oral presentation.

Viaene, J., Willekens, K., Vandecasteele, B. and Reubens, B., 2015. Boerderijcomposteren in samenwerkingsverband? CriNglooP Collectief, Merelbeke, Belgium, 8/10/2015. Poster presentation.

Viaene, J., Reubens, B., Vandecasteel,e B., Willekens, K. and De Neve, S., 2013. Use of by-products as system innovation: GeNeSys: Closing cycles trough composting. 18th National Symposium on Applied Biological Sciences, Ghent, Belgium, 8/02/2013. Poster presentation.

Scientific reports

Nelissen, V., Viaene, J., Reubens, B., Vandecasteele, B. and Willekens, K., 2015. Optimaliseren van de opslag en bewerking van runderstalmest op de kopakker. ILVO-studie met financiële ondersteuning van Boerenbond, 49p.

Viaene, J., Nelissen, V., Reubens, B., Vandecasteele, B. and Willekens, K., 2015. Optimaliseren van de opslag en behandeling van vaste rundermest via compostering of inkuilen. Bijlage: Effect van eindproducten op N mineralisatie en gewasopbrengst van Engels raaigras. ILVO-studie met financiële ondersteuning van VCM, Boerenbond, Agriton en Orffa, 9p.

Viaene, J., Nelissen, V., Reubens, B., Vandecasteele, B. and Willekens, K., 2015. Optimaliseren van de opslag en behandeling van vaste rundermest via compostering of inkuilen. Procesverloop en productkwaliteit. ILVO-studie met financiële ondersteuning van VCM, Boerenbond, Agriton en Orffa, 77p.

Viaene, J., Reubens, B., Vandecasteele, B. and Willekens, K., 2014. Composteren als valorisatievorm van reststromen in de Vlaamse land- en tuinbouw: Knelpunten en opportuniteiten. ILVO Mededeling nr. 167, 61p.

Gybels, R., Viaene, J., Vandervelden, J., Reubens, B. and Vandecasteele, B., 2013. Biomassa als bodemverbeteraar - Onderzoek naar de toepassing van beheerresten als bodemverbeteraar. Agentschap voor Natuur en Bos, Inverde & ILVO, 73p.

Chapters in scientific reports

Reubens, B., Willekens, K., Vandecasteele, B., Nelissen, V. and Viaene, J., 2014. Naar een optimalisatie van de opslag en behandeling van kippenmest en vaste rundermest via compostering. In: Ruysschaert, G., Vandecasteele, B., Willekens, K., Van Waes, J., Van Laecke, K. (eds.) Bodem, Nutriënten en Compost: Onderzoek voor een Duurzame Landbouw. ILVO Mededeling, nr. 171, Instituut voor Landbouw- en Visserijonderzoek, p. 200-215.

Viaene, J., Reubens, B., Willekens, K. and Vandecasteele, B., 2014. GeNeSys: Valorisatie van groene en bruine nevenstromen op het landbouwbedrijf. In: Ruysschaert, G.,

Vandecasteele, B., Willekens, K., Van Waes, J., Van Laecke, K. (eds.) Bodem, Nutriënten en Compost: Onderzoek voor een Duurzame Landbouw. ILVO Mededeling nr. 171, Instituut voor Landbouw- en Visserijonderzoek, p. 216-223.

Willekens, K., Reubens, B., Vandecasteele, B. and Viaene, J., 2012. Waarde van compost voor de bodemvruchtbaarheid. In: De Cock, L., Van Waes, J. (eds.) De Biologische Landbouw in Vlaanderen: Onderzoek 2011-2012. NOBL - Netwerk Onderzoek Biologische Landbouw en Voeding, p. 34-35.

Supervision of bachelor and master students and internships

Clodualdo Capito, 2015. N mineralization and N_2O emission following incorporation of fresh and processed vegetable crop residues. Master of Science in Physical Land Resources, UGent-VUB. Supervisor: Prof. Dr. ir. Stefaan De Neve.

Carmen Molina, 2014. Internship 'Composting solid manure', funded by the Leonardo da Vinci Programme of the European Commission's Lifelong Learning Programme.

Cyril Amelynck, 2013. Valorisatie van oogst-en beheerresten. Bachelor in de Agro- en biotechnologie, Afstudeerrichting Groenmanagement, Technologiecampus Gent, KU Leuven. Supervisor: ir. Erik De Boodt.

Ginés López, 2013. Internship 'Composting crop residues', funded by the Leonardo da Vinci Programme of the European Commission's Lifelong Learning Programme.

Peer reviewing

Peer reviewing for 'Waste Management' (1)

Specialist courses (Ghent University)

2014: Effective graphical displays

2013: Authentic networking

2013: Analysis of Variance

2013: Advanced academic English: Writing skills

2012: Conference skills